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Alpine Waters

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Aims and Scope

Since 1980, *The Handbook of Environmental Chemistry* has provided sound and solid knowledge about environmental topics from a chemical perspective. Presenting a wide spectrum of viewpoints and approaches, the series now covers topics such as local and global changes of natural environment and climate; anthropogenic impact on the environment; water, air and soil pollution; remediation and waste characterization; environmental contaminants; biogeochemistry; geoecology; chemical reactions and processes; chemical and biological transformations as well as physical transport of chemicals in the environment; or environmental modeling. A particular focus of the series lies on methodological advances in environmental analytical chemistry.

Series Preface

With remarkable vision, Prof. Otto Hutzinger initiated *The Handbook of Environmental Chemistry* in 1980 and became the founding Editor-in-Chief. At that time, environmental chemistry was an emerging field, aiming at a complete description of the Earth's environment, encompassing the physical, chemical, biological, and geological transformations of chemical substances occurring on a local as well as a global scale. Environmental chemistry was intended to provide an account of the impact of man's activities on the natural environment by describing observed changes.

While a considerable amount of knowledge has been accumulated over the last three decades, as reflected in the more than 70 volumes of *The Handbook of Environmental Chemistry,* there are still many scientific and policy challenges ahead due to the complexity and interdisciplinary nature of the field. The series will therefore continue to provide compilations of current knowledge. Contributions are written by leading experts with practical experience in their fields. *The Handbook of Environmental Chemistry* grows with the increases in our scientific understanding, and provides a valuable source not only for scientists but also for environmental managers and decision-makers. Today, the series covers a broad range of environmental topics from a chemical perspective, including methodological advances in environmental analytical chemistry.

In recent years, there has been a growing tendency to include subject matter of societal relevance in the broad view of environmental chemistry. Topics include life cycle analysis, environmental management, sustainable development, and socio-economic, legal and even political problems, among others. While these topics are of great importance for the development and acceptance of *The Handbook of Environmental Chemistry,* the publisher and Editors-in-Chief have decided to keep the handbook essentially a source of information on "hard sciences" with a particular emphasis on chemistry, but also covering biology, geology, hydrology and engineering as applied to environmental sciences.

The volumes of the series are written at an advanced level, addressing the needs of both researchers and graduate students, as well as of people outside the field of "pure" chemistry, including those in industry, business, government, research establishments, and public interest groups. It would be very satisfying to see these volumes used as a basis for graduate courses in environmental chemistry. With its high standards of scientific quality and clarity, *The Handbook of*

Environmental Chemistry provides a solid basis from which scientists can share their knowledge on the different aspects of environmental problems, presenting a wide spectrum of viewpoints and approaches.

The Handbook of Environmental Chemistry is available both in print and online via www.springerlink.com/content/110354/. Articles are published online as soon as they have been approved for publication. Authors, Volume Editors and Editors-in-Chief are rewarded by the broad acceptance of *The Handbook of Environmental Chemistry* by the scientific community, from whom suggestions for new topics to the Editors-in-Chief are always very welcome.

Damià Barceló
Andrey G. Kostianoy
Editors-in-Chief

Volume Preface

Mountains are the water towers of the world. However, they are not only an important source of water but also a source of key resources such as energy, minerals, and forest and agricultural products. They offer a multitude of recreational opportunities, and probably most importantly, serve as storehouses for biological diversity. Mountains provide the living space for about 10% of the world's population. A much larger percentage of the population profits from the various services of mountains, most of which depend on water.

Mountains play a key role in the global water cycle as well as in numerous regional water cycles. Mountain waters often are subject to a manifold of conflicting interests and pressures. Because of growing populations and needs for economic development, water demands for humans, food, energy, and industry will certainly increase. Furthermore, climate change will lead to extended periods of drought, higher flood risks, altered physical, chemical, and biological properties of water, and to increased demands for irrigation. At the same time, the ecological functioning of our waters must be maintained and improved. As a consequence, the diverse pressures on mountain waters will likely rise dramatically in the future. Aiming towards sustainable mountain water management is a highly significant and important future challenge.

The aim of this book is to extensively portray the highly diverse attributes of mountain waters. With this aim, we hope to not only convey a sound scientific insight but also to demonstrate the paramount importance of mountain waters for future ecological and societal development.

The book starts off with a synthesis on mountain water features and management concerns. This chapter summarizes and complements the contents in the following chapters, and hence aims at transmitting a comprehensive view on the diverse mountain water issues, in general. The book is then divided into four parts comprising 13 chapters:

Part I, *Alpine Water Resources*, examines the hydrological basics, the impacts of climate change in the Swiss Alps, and human interventions in mountain waters.

Part II, *Biogeochemistry and Pollution of Alpine Waters*, deals with the chemistry of mountain rivers, the effects of acid deposition on high elevation lakes, the glaciers as archives of atmospheric deposition, and the occurrence of persistent organic contaminants.

Part III, *Ecology of Alpine Waters*, discusses the ecological relationship between different water sources and associated habitats, important abiotic factors, and the biology of alpine streams.

Part IV, *Case Studies*, presents four studies on integrated water assessment and management.

The discussion of important scientific basics is supplemented with considerations on the various uses of mountain waters, needs for management actions, and on future challenges towards sustainable water management. This overview is set not only on mountain areas themselves, but also on downriver reaches and the surrounding lowlands, and hence on the relationship between mountain and lowland water issues. The book has a clear focus on the European Alps but some chapters refer to mountains on other continents. Most of the generalities regarding the natural processes governing mountain waters, on conflicting water uses, and on management needs will be universally valid. This also holds true for the three case studies elaborated in Switzerland (Part IV) whose findings and insights are certainly of general significance. A further case study examines a highly challenging water situation in the Middle Mountains watersheds in the Himalayas.

Elaboration of the book has turned out to be a demanding and, at the same time, productive task for the contributing editors, authors, and editorial assistants. The chapter contributions demonstrate the high competence and great passion of the authors regarding mountain waters. The outstanding collegial spirit in the preparation of the book resulted in a delightful experience. To all who have contributed to the book, I express my warmest thanks and my great appreciation.

Dübendorf, Switzerland
September 2009

Ulrich Bundi
Principal editor

Contents

Synthesis: Features of Alpine Waters and Management Concerns

Ulrich Bundi

Abstract This chapter aims at facilitating a comprehensive understanding of the alpine water issue. The nature and problems of alpine waters comprise a wide spectrum of natural features and processes, uses, pressures, and management options. Many of these aspects are presented in the following Chapters, but obtaining a comprehensive synthesis is certainly not an easy task. The content of the volume Chapters is synthesized and complemented with some basic considerations regarding water assessment, rising water conflicts and management issues. On the basis of this background, the necessity of integrated water management and inter- and transdisciplinary cooperation is emphasized. A vision of sustainable water management is presented towards supporting future orientations for water policy and management.

Keywords Alpine, Ecosystem services, Human uses, Integrated water management, Water vision

Contents

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1 Mountains: General Characterization

Mountains are landforms that extend above the surrounding landscape. Definitions of mountains depend on what aspects are considered. A concise, comprehensive description is given by Goudie [1] who defines mountains as "substantial elevations of the Earth's crust above sea level which result in localized disruptions to climate, drainage, soils, plants and animals."

Mountain areas typically show great variation in topography, geology, climate and biological zones. Besides natural variation, human activities such as agriculture, tourism, industry and trade can be highly variable. Heterogeneous natural conditions and near-natural agricultural practices provide a wide spectrum of habitats for fauna and flora. Numerous areas of high species diversity, so-called "biodiversity hotspots," are found in the Alps and mountain regions in general. They are storehouses of biological diversity, holding endemic and endangered species.

The Agenda 21, endorsed at the Earth Summit 1992 in Rio de Janeiro, states: Mountains are an important source of water, energy and biological diversity. Furthermore, they are a source of such key resources as minerals, forest products and agricultural products and of recreation. As a major ecosystem representing the complex and interrelated ecology of our planet, mountain environments are essential to the survival of the global ecosystem. Mountain ecosystems are, however, rapidly changing. Mountains are highly vulnerable to human and natural ecological imbalance. Mountains are the areas most sensitive to all climatic changes in the atmosphere. About 10% of the world's population depends on mountain resources. A much larger percentage draws on other mountain resources, including and especially water [2].

2 Mountain Waters: Factors and Features

Compared with surrounding lowlands, mountain regions are often rich in water. Air masses flowing towards the mountains are forced to rise and subsequently cool, which leads to the release of humidity in the form of precipitation. Discharge from mountain areas can be highly disproportionate to the total discharge of a catchment. For large rivers originating in the Alps, this disproportion factor equals 1.5–2.6 (share in annual catchment discharge vs. share in total area, cf. Sect. 5.2 in [3], this volume). Therefore, most mountain regions play an important role as water towers for the lowlands. It was found that more than 50% of mountain areas play an essential or at least supportive role for the water resources in the respective downstream regions (cf. Sect. 5.3 in [3], this volume).

Precipitation shows great intramountain variation. Determining factors comprise windward and leeward orientation to prevailing wind directions and the effects of local wind systems that depend on physical processes and relief. In general, annual amounts of precipitation tend to increase with altitude. Precipitation can occur in solid or liquid form. Further, rain fog can substantially contribute to liquid precipitation locally. Snow is the dominant form of solid precipitation. The temporary accumulation of snow during winter plays an important role in the alpine water balance (cf. Sects. 2 and 3 in [3], this volume).

The water of mountains appears in different forms, comprising rivers and streams, different kinds of natural lakes, artificial reservoirs, groundwater, and snow and glaciers. Often, these different water forms represent distinct elements in mountain landscapes. These various kinds of waters merge forming longitudinal chains or networks, and interlinking regions and countries.

Alpine streams, themselves, consist of many different types: quiet springs, mellow lake outlets, steep whitewater torrents, meandering and braided stream channels in flood plains, roaring waterfalls, and glacier streams that rage at certain times and are calm at others. Alpine streams also flow over artificial steps, have channels confined by man-made banks, or even stream stretches suffering from total withdrawal of water [4]. Alpine streams are classified according to their dominate source of water: glacier-fed (*kryal streams*), groundwater-fed (*krenal streams*), snowmelt-fed (*rhithral streams*). These water sources vary over temporal scales that range from interannual to diurnal. The source of water plays an important role in influencing hydrology, water temperature, channel stability, sediment regimes, and water chemistry. In winter, ice formation and snow cover can have a major influence on habitat conditions. Ice formation and ice transport can profoundly affect river hydraulics and morphology, and cause high mortality of aquatic biota (cf. Sects. 1–4 in [5], this volume). Climatic and topographic barriers lead to fragmentation, creating an array of habitats and biotic communities [6]. The different factors influencing flow and morphology add up to distinct habitat properties that are a major determinant of biotic community structure (cf. Sects. 5–8 in [5], this volume).

Glacier-fed streams (kryal) have maximum flow discharge during maximum snowmelt. Water temperature is low, typically $< 4.0°C$, especially below the glaciers where it averages $< 1.0°C$. Turbidity and suspended sediment levels are very high during ice-melting periods. *Groundwater-fed streams* (krenal) have relatively constant discharge and water temperature, stable channels, and low turbidity. In *snowmelt-fed streams* (rhithral), discharge is composed of snowmelt and runoff. Flow and temperature regimes are the most variable among stream types. Sediment transport can be high during the snowmelt period and floods from rainfall events. *Combinations of source-specific characteristics* exist the farther away a stream section is from a dominant water source (cf. Sects. 2, 3, 5 in [5], this volume).

The chemical composition of rivers in the Alps is strongly regulated by natural diffuse inputs originating from rock weathering. Weathering mainly occurs as an interaction of water with rocks. Physical weathering is strongly influenced by frost. It leads to solid particles ranging from boulders to silt that result in increased surface area of the solid matter and enhanced chemical weathering. Smaller particles and dissolution processes are dominant sources of diffuse inputs. Human activity generating wastewater or diffuse inputs through agricultural activity can directly or indirectly influence chemical water composition. The effects of anthropogenic pollution loads are governed by dilution, which is often high but decreases gradually in the lower reaches of the watershed (cf. Sect 1.2 in [8], this volume).

Biological features (cf. Sects. 7 in [5], this volume, 1–4 in [9], this volume): Autochthonous benthic algae are the primary food resource for benthic invertebrates in Alpine waters. Higher aquatic plants are essentially lacking, although mosses can be found in groundwater-fed streams. The complete algal diversity of Alpine waters is poorly known. Focus often is on diatoms whose communities are rich in taxa. Alpine waters offer suitable environmental conditions for endangered or extremely rare diatom taxa by playing an important role as refugia. Zoobenthos of surface waters show substantial spatial and temporal variation, and the spatial heterogeneity of surface waters may enhance their overall biodiversity. In general, aquatic insects comprise a substantial proportion of the zoobenthos in surface waters, with Chironomidae being most common. The abundance and diversity of zoobenthos typically increases downstream as stream environments become more physically favorable, for example more stable, higher temperatures. The Diamesa-groups (chironomids) usually dominate upper glacial streams, with Ephemeroptera (*Baetis alpinus* and *Rhithrogena* spp.), Plecoptera, and Trichoptera becoming more common in lower sections, and also in groundwater streams. Lake outlets provide another habitat type that often is inhabited by a distinct assemblage of zoobenthos. Fishes in Alpine waters are limited to cold stenothermic species such as the brown trout. Water abstraction and flow regulation severely constrain the management of the fishery in Alpine waters today.

High Alpine lakes are extreme ecosystems that are shaped by harsh climatic conditions, scarcity of nutrients, and low salinity. They are not, however, too remote to receive inputs of anthropogenic chemicals. Air pollutants such as sulfur dioxide, nitrogen oxides and ammonia stemming from densely populated and industrialized areas are transported via the atmosphere and deposited in high alpine areas. They may lead to acidification of soils and lakes, if they exhibit a low buffering capacity, and have negative effects on water biology (cf. Sect. 5 in [7], this volume). Because of the cold-trapping effect, considerable quantities of organic chemicals are deposited at high altitudes and lead to increased concentrations of persistent chemicals in alpine lakes and aquatic organisms. (cf. sect. 7 in [10], this volume). In respect to fishes, many Alpine lakes are currently stocked to sustain the fishery (cf. Sect. 4.1 in [9], this volume).

Climate change will likely be an important driver for changes of basic features of Alpine waters. Recession of glaciers and snowline increase will markedly influence flow regime and sediment dynamics of streams. More and more waters will likely shift from glacier- to snow-dominated ecosystems. Glacier retreat may promote the enlargement or formation of proglacial lakes (cf. Sect. 5.3 in [9], this volume).

3 Functions of Mountain Waters

Mountain water resources provide vital services for nature, the economy, food production, and human health and well-being. Mountain water functions comprise a wide spectrum from economic uses like energy production, life support such as providing habitat for plants and animals and drinking water supply to recreational activities and emotional enrichment. A different dimension of water is its potential to cause floods and landslides, thus posing a serious risk to human life and economic values. As the water supplier for lowland areas with high population density, mountain water resources clearly play a pivotal role (cf. Sect. 5.3 in [3], this volume).

Hydropower production is an especially important factor in many mountain regions. High water availability and large differences in altitude over short distances offer highly favorable conditions for hydropower. In the Alpine countries, the importance of hydropower is greater the higher the proportion of the Alpine area. In the case of Switzerland and Austria, hydropower amounts to 57% and 59%, respectively, of the total electrical production. In the Swiss Alps, the usable hydropower production potential is largely realized (cf. Sect. 2.2 in [11], this volume).

Water resources are used for *irrigation* in a different spatial context. First, they are used for irrigation of inner mountain lands, which has a long tradition in the Alps and particularly in its central dry valleys. The water is usually abstracted high in the mountains and often piped across extremely difficult terrain to lower fields and pastures along the valley floor. Other examples are the Middle Mountains watersheds in the Himalayas. In these densely populated areas food supply is based on subsistence agriculture carried out on irrigated terraces. High population growth and climate change will enormously heighten the pressures on water use and management. Maintaining and improving the complex production and irrigation systems will be very demanding tasks (cf. [12], this volume). Second, especially in mountains with arid or semiarid surroundings, numerous reservoirs have been built with the purpose of delivering irrigation water to agricultural land in the lower catchment or in regions outside the natural catchment. In these cases, water often is transported in long channels and pipes to the areas where it is needed.

Other important uses of mountain water resources are the *supply of drinking and process water* for towns, tourist centers and industry. Thereby, *wastewater* arises that is discharged into receiving waters, often after collection, transport and treatment in a sewage system. Hence, rivers and streams serve as transport media for the pollution load of wastewater, and as dilution and self-purification media. Water use for *artificial snow making* for ski slopes has been strongly promoted in the Alps in the last 10–20 years. Even though the influence of this use on the overall water cycle is of secondary importance, local shortages and water conflicts are likely to occur (cf. Sect. 2.3 in [11], this volume).

Mountain waters offer a multitude of opportunities for *recreational and sporting activities*. They enrich the landscape, are essential for human well-being, and are often the source of strong emotions. A diverse landscape and intact streams increases the recreational value of a region. One use of Alpine streams that is no longer practiced is the *drifting or floating of logs*, although it was common well into the twentieth century. This practice often required hydrologic engineering, such as retaining basins or sluices for directing logs [13].

4 Anthropogenic Impacts on Water Resources

Anthropogenic impacts occur through direct interference of waters or indirectly as a consequence of different kinds of non water-related activities. Among direct interferences, correction works, hydropower generation, water withdrawal for irrigation, domestic and industrial water uses, and the discharge of treated and untreated wastewater are most important. Manifold direct impacts can also arise from recreational activities like boating and intensive bathing activities.

The production of *hydroelectric power* requires a number of different production and storage structures resulting in operational and structural impacts that affect streams in a variety of ways (Table 1) (cf. Sect. 2.2 in [11], this volume) [4, 14]. The groundwater regime often is affected by hydropower plants through interventions of infiltration and exfiltration zones. Storage basins often receive water from areas that do not belong to the natural drainage area. Water is transferred within a river watershed or even between watersheds of different river systems. In the Swiss Alps, virtually all large and many small streams are affected by water withdrawal and/or hydropeaking.

Dams and reservoirs have been built in the last 100 years in vast numbers throughout many mountain areas. They serve different purposes, mainly irrigation, hydropower, water supply or flood control, or they aim at pursuing multipurpose functions. Effects on flow regime, for example shifts in summer and winter patterns, on sediment dynamics, and on other features of the aquatic ecosystems can be dramatic (cf. Sects. 2.2 in [11], this volume, 2 in [15], this volume).

Correction works: A large number of rivers and streams have been corrected in the Alps with the aims of protecting settlements from flooding, reclaiming farm land, and reducing epidemic plagues. Correction works comprise straightening of the stream course, riverbed stabilization, levees and different kinds of other measures. (cf. Sects 2.4 in [11], this volume, 2 in [16], this volume) Because of extensive correction works in the Swiss Alps, only a few relics of formerly widespread flood plains remain today.

Wastewater is predominantly discharged into running waters. An increase in the concentrations of organic and inorganic nutrients (C, P, N), metals and chemicals

Table 1 Primary P and secondary effects S of hydroelectric power generation on streams. S are a consequence of P

Physical and chemical effects of hydropower utilization	Type of interference				
	Water withdrawal	Sand removal (sand flushing)	Water return	Water storage	Structural interventions
Changes in discharge regime	P	(P)	P		
Changes in flow pattern	S		S		P
Changes in suspended solid loads	S	P	S	P	(S)
Shrinking and/or structural changes in habitat	S	S	S		P
Changes in chemistry and temperature of water and sediments	S		S	P	

depends on the degree of treatment and on the dilution rate. In the Alps, wastewater treatment is common, and dilution often high. However, in many areas with large tourist centers, water pollution abatement requires advanced and costly measures.

Non water-related activities and their impacts on waters occur at different spatial scales, from local and regional to global. They comprise activities in agriculture, forestry management, urban and road development, and different kinds of combustion processes in housing, industry, energy production and traffic. An example of a local context is nitrate accumulation in groundwater because of intensive farming. The presently best known global effect is climate change and subsequent hydrological alteration as a consequence of worldwide combustion of fossil fuels.

Acidification pollutants, *nutrients* and *chemicals* being transported via the atmosphere to alpine regions originate from both nearby and far away sources. Deposition mainly occurs on the land surface where they can lead to soil and subsequent water acidification. Nutrients and chemicals are partly released from soil and land surfaces to water depending on their chemical nature, and on land and soil characteristics (cf. [7, 10], this volume).

Alpine areas are highly sensitive to *climate change*, much more than other European regions. Effects comprise an increase in temperature, and a change in the quantity, intensity and seasonality of precipitation. In the Alps, an increase in winter precipitation is likely to occur in the coming decades. Other changes include a decrease in summer precipitation, increased evapotranspiration, an increase of snowline with altitude, a dramatic reduction of glaciers, more frequent extreme precipitation events and higher flood risks. These changes will lead to problems of water availability within the Alps and more so in downstream regions (cf. [17], this volume).

5 Ecological Effects of Anthropogenic Interventions

In a holistic view, the condition of a water body is given by the totality of all physical, morphological, chemical and biological factors that influence and characterize the water habitat, and which are important with regard to human water uses. However, the condition of waters can only be characterized by a selected set of measurable parameters and a description of observed phenomena (Fig. 1). For biological features, abiotic factors play a pivotal role. Therefore, alterations of abiotic factors can lead to changes in biological features. Table 2 gives a synopsis on the ecological effects of different human interventions.

Hydrological interventions like water withdrawal, water return and water storage directly change discharge regimes, and this affects various abiotic factors. Habitat features will be altered, and longitudinal, lateral and vertical connectivity may become disturbed or even disrupted, thus leading to habitat fragmentation. Analogous effects can result from indirect interventions by land-use change and climate change. Biological changes can be drastic on both single species as well as on the total biocenosis.

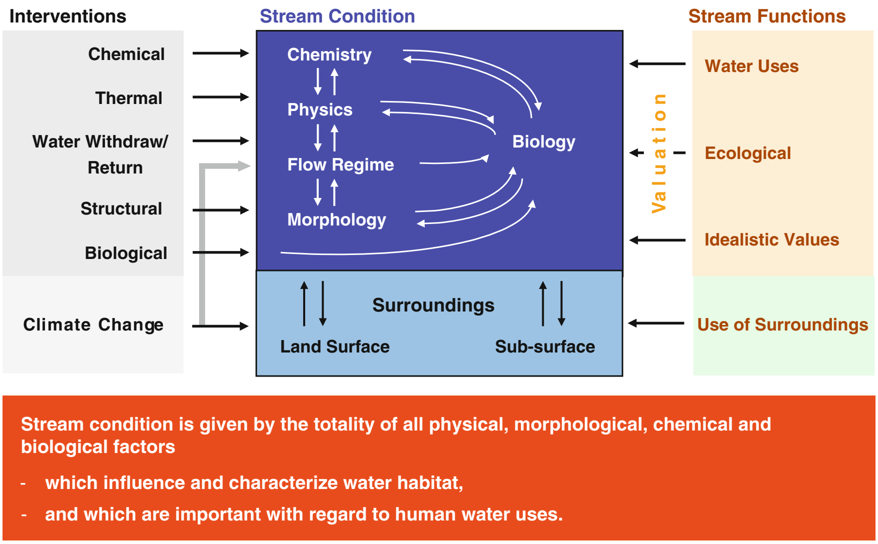

Fig. 1 Factors determining the condition of waters

Morphological interventions primarily comprise weirs, dams and other river correction works. Similar types of abiotic and biological effects like the ones from hydrological interventions can occur.

Thermal interventions occur for diverse reasons. Direct heating occurs from wastewater and cooling water discharge and leads to temperature increases. Probably more important are indirect warming and cooling effects from hydrological and morphological interventions (Sects. 2.2.4 in [11], this volume, 4 in [15], this volume). Temperature sets limits on where a species can live, whereas maximum summer temperature is more critical than low temperature extremes, for example for brown trout. Changes in temperature regime can also accelerate or delay reproduction cycles of insects and therewith endanger population survival.

Chemical interventions comprise adding C-, P, and N-compounds from wastewater and agriculture, acidifying compounds transported via the atmosphere, toxic substances like nitrite and heavy metals, and chemicals. Depending on the type and loads of pollutants, they can lead to eutrophication, to an increase in acidifying potential, to increased concentrations of toxic compounds, or to the appearance of synthetic chemicals that do not occur naturally. Eutrophication results in mass growth of heterotrophic and phototrophic organisms, oxygen depletion, and reduction or eradication of oxygen-sensitive species. Natural-type toxic compounds and chemicals can lead to acute and chronic toxic effects of a wide variety. From these effects, single organisms as well as whole communities can be severely affected, and far-reaching consequences for aquatic life are possible.

Understanding of the processes governing ecological effects is limited, especially concerning the combinations of different types of interventions. Under different mountain conditions, different kinds of interventions and effects may be dominant.

Table 2 Ecological effects of anthropogenic interventions

Anthropogenic interventions		Abiotic effects/changes Habitat alteration		Biological effects
Hydrological – Water withdrawal – Water storage – Water return – Hydropeaking – Land use change (indirect) – Climate change (indirect)	⇨	Δ Discharge regime Δ Flow pattern Δ Turbidity/sediment transport Δ Temperature regime + Longitudinal disruption + Lateral disruption + Vertical disruption → Destruction of ecotones → Δ Instream structures → Δ Habitat features → + Habitat fragmentation	⇨	▪ Elimination of species ▪ Disturbing or making impossible reproduction ▪ Disturbing interactions between organism ▪ Disturbing or eliminating biological interactions between aquatic river stretches, and between surface and groundwater ecosystems, and between aquatic and terrestrial ecosystems ▪ Change of organism occurrence and diversity
Morphological – Weirs and dams – Stream course straightening – Bottom stabilization – Bank stabilization – Artificial steps – Levees	⇨			
Thermal – Heat insertion – Warming and cooling effects from hydrological and morphological interventions (indirect) – Climate change (indirect)	⇨	Δ Temperature regime + Temperature – Temperature	⇨	▪ Elimination of species ▪ Disturbance of reproduction cycles ▪ Change of organism occurrence and diversity
Chemical – Waste water discharge – Agriculture (indirect) – Atmospheric input (indirect) – Chemical effects from hydrological and morphological interventions, and from temperature change (indirect)	⇨	+ Nutrient concentrations (C, P, N) → Oxygen depletion + Acidifying compounds → Acidification + Toxic compounds (e.g. nitrite, metals) + Chemicals	⇨ ⇨	▪ Mass growth of hetero- and phototrophic organism ▪ Elimination of species ▪ Acute and chronic toxic effects ▪ Change of organism occurrence and diversity

In the case of the Alps, the majority of negative ecological effects are caused by hydropower generation. River corrections with negative ecological impacts are mainly located in densely populated and highly used inner mountain valleys. Biological effects due to chemical pollution are mostly restricted to river stretches downstream of the pollution source.

6 Rising Pressures on Mountain Water Resources

Demands on mountain water resources are driven by local, regional or supra-regional interests and the different interests can be in conflict. For example, hydropower generation and river correction are often in sharp conflict with the desire for an intact, natural landscape, recreation and tourism. Further, discharge of wastewater into the natural water cycle can endanger or even make drinking water use impossible.

Lowland regions, on the one hand, depend on electricity and on water from mountain regions for irrigation, human use and industry. On the other hand, water management in the mountains can affect flow regime and water quality of downstream rivers and as such be in conflict with downstream uses and conservation objectives. Examples of downstream interests include improvement of river habitat, flood protection, navigation and water supplies. Downstream water users are becoming more sensitized to these issues, especially when considering the possible consequences of global climate change. In the future, water management in the mountains will be confronted more with the problems and demands of downstream regions. However, these regions also have some responsibilities of their own. The farther they move from a sustainable use of their own water resources, the more they will depend on the import of water and on sound water management in mountain areas [18]. All of this demonstrates rather clearly how the different interests regarding water resources are intertwined, from the mountains all the way to the sea.

Pressures on water resources are already high, but they will further increase in the coming years and decades. Different kinds of pressures and driving forces can be identified [19]:

Hydropower generation: Considering worldwide limited energy resources and the need for $C0_2$-reduction, hydropower will be further promoted. Looking at the great potential of mountain areas, pressure on building new hydropower plants and, hence, on the ecological integrity of water resources will grow enormously.

River correction and rehabilitation: Intensive use of the living space and often one-sided use priorities that disregard ecological values are the major reasons for the numerous river corrections in the Alps, often causing serious physical impairment. Mainly in low lying inner mountain valleys of the Alps and further mountain areas, pressure for river correction will still increase. Population growth and subsequent need for urban development and expansion of economic activities as well as climate change will lead to higher flood risks. However, in the past 20 years a paradigm change concerning river correction has gained wide acceptance. In the future, a focus will lie on creating preconditions for near natural water cycles, on providing sufficient space for sound river development, on environmental soundness of engineering works, and on hydrological optimization of reservoirs. Furthermore, emphasis will be placed on the rehabilitation of impaired rivers.

Agriculture will have to increase its production capacities in order to meet local, regional and global food demands of growing populations. Because of their water richness, mountains and their surrounding areas are well-suited to expand agricultural activities. Subsequently, both pressure on withdrawing water resources for irrigation and water pollution loads through agricultural activity will substantially grow.

Urban development will be seen as further growth of agglomerations, peri-urban sprawl and the depopulation of peripheral regions. In growth regions, pressure on the land adjacent to rivers and flooding risk will increase, and the functions of the land for supporting near-natural water cycles will be disturbed. In peripheral regions, risks might arise from the weakening of management structures needed for maintaining proper wastewater disposal on the one hand, while chances might develop for creating new natural areas on the other.

Urban water management which comprises a wide spectrum of technical, operational, organizational and financial features is facing great challenges. Further development in management is an indispensable prerequisite for keeping waters in a healthy state. Enormous costs for the maintenance and technical/ecological improvement of infrastructure must be secured. Securing the necessary financing, development of effective organizational forms and implementing up-to-date scientific and technical competence will be difficult and complex.

Recreation and tourism are of growing importance in many mountain regions. They are, to a good part, based on multifaceted intact landscapes in which waters clearly play an important role. Hence, waters should convey intact natural sceneries on the one hand, while increasingly experiencing stress from recreational activities.

Chemicals that are used in industry, trade, households, the health sector, traffic and agriculture represent an ongoing and probably growing complex of problems. Many chemicals find their way into the environment and the water cycle. Even though, they usually occur only as trace amounts, antibiotics, environmental hormones, pesticides, and many other chemicals can be harmful to aquatic organisms and humans. Various countries and international organizations have introduced regulations governing the use of chemicals. Nevertheless, detection of chemicals and metabolites as well as their biological effects has turned out to be highly demanding. Until now, the state of knowledge and mitigation measures can not keep pace with the development of chemical problems.

Climate change will lead to an increase of snowline with altitude, extended summer drought periods in inner Alpine valleys, to problems with water availability in downstream regions and to higher flood risks. Hence, new challenges for water supply, agricultural production and irrigation, and for flood protection will arise. Competition between different water uses will be heightened.

Ecological function and biodiversity will in the future gain more attention. Aquatic organisms and their communities depend on healthy living conditions. Intact water networks play an important role for preserving and sustaining biodiversity of waters and landscapes. The ecological integrity of waters is a precondition for many ecosystem services that benefit humans such as the supply of clean water and provision for recreation opportunities.

International obligations of mountain countries will be of high relevance in the future. As stated at the beginning of this section, upstream regions will increasingly be challenged by claims from downstream regions for sound mountain water management, and for providing irrigation and drinking water.

7 Need and Opportunities for Action

The waters of the mountains serve both nature and society. One-sided use priorities that disregard the values of a natural mountain environment or the interests of neighboring regions must be rejected. The need for action is given at local, regional and international levels, and in the interaction between levels. In the past, focus was on the prevailing challenges in the sectors of "water utilization," "water protection," and "flood protection."

The overall goal is sustainable water resources management, which comprises two basic thrusts: first, integrated water management aiming at effective and efficient protection and use concepts for water; second, environmentally sound rearrangement of human activities such as energy production and use, the kinds of habitation, traffic, industrial production and agriculture. The ecological reorientation of activities is indispensable not only for water but for other reasons such as energy supply and climate change, and ultimately necessitates basic societal changes towards sustainable development.

7.1 Vision of sustainable water management [20]

The use and protection of water resources are key factors for a flourishing society, the economy, and the environment. Mountain countries have an obligation to maintain good water quality and a natural discharge regime that supports various water uses and flood protection for downstream regions. The water management sector is guided by the principles of sustainable development. These principles aim to optimize value creation in the long term in the management of water for society and the environment. Risks and threats related to water dynamics must be averted or mitigated as effectively as possible.

Uses to be permanently assured are:

- *All of the essential uses of water that include water supply, food production, human and animal health, energy supply and transport, and thereby the economy, prosperity and well-being of societies.*
- *Protection against flooding so as to preserve human life, usable land and assets.*
- *Ecological functions of aquatic ecosystems as habitats for self-regulating biological communities, and as elements in ecological networks that form landscapes.*
- *Recreational functions, aesthetics, and the emotional value of surface waters.*

Conflicts of interest among these fundamental concerns are solved from an integrated perspective. Water resources are managed on a cross-sectoral and mainly regional basis, if possible taking the hydrological catchment as the reference area. Management must be adapted to natural conditions, and the spatial planning, and economic and social aspects of the region.

Water resource management must be developed using transparent procedures involving all key interests and stakeholders. In spatial planning, the concerns of sustainable water management are coordinated with other important areas of action. In this process, accountability is taken across supra-regional, national and international interests of water resources. The overall framework for the management of water resources creates incentives for all actors to take responsible measures at their own initiative. Much weight is given to promoting technical expertise on the part of all actors, as well as an understanding of water in society.

Science is challenged to decisively support the path towards sustainable water management. On the one hand, research aimed at understanding the processes governing mountain water ecology and water use must be strengthened. On the other hand, science needs to assist in the development and testing of integrated management approaches, in particular when relevant political, legal, economic, institutional, social and cultural aspects are considered. This approach can only be done in strong partnership with the relevant water stakeholders. For science, this means playing a novel role, one which will be most demanding.

In Part IV of this volume four cases of water assessment and management adopting an integrated approach are presented. First, the development and implementation of an eco-label for hydropower generation is described. The label considers the regional societal pressures and aims at mitigating ecological impacts (cf. [21], this volume). The numerous reservoirs in mountain areas can have dramatic effects on flow regime and other features of the aquatic ecosystems. In [15], this volume, a comprehensive analysis of the effects and the mitigation options of reservoir management is presented. In consideration of the widespread negative ecological impacts of river correction works, emphasis will have to be placed on restoration attempts. In the Rhone-Thur study basic principles of river restoration have been elaborated (cf. [16], this volume). Sustainable agriculture and water management will be crucial for coping with the challenges of population growth and climate change in the Middle Mountains watersheds in the Himalayas. Maintaining and improving the complex production and irrigation systems will be very demanding tasks (cf. [12], this volume).

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Part I
Alpine Water Resources

General Characteristics of Alpine Waters

Bernhard Wehren, Rolf Weingartner, Bruno Schädler, and Daniel Viviroli

Abstract The elements of the water balance, namely precipitation, runoff, evapotranspiration, and storage change, their interaction and special attributes in the mountains are presented using the example of the European Alps, with particular reference to Switzerland. Strong differentiation in the alpine climate over time and space exerts a significant influence on the water cycle. This chapter therefore discusses each of the elements of the water balance with particular reference to the influence of mountains and their measurement, as well as the spatial differentiation characteristics. The analysis of the water balance is accompanied by a discussion on the attributes and differences at different altitudes and in different climatic regions. Finally, the importance of alpine water resources for water supplies in the adjacent lowlands is examined.

Keywords Alpine hydrology, Precipitation, Runoff, Water balance, Water towers

Contents

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1 Introduction

Mountains play a special role in the hydrological cycle of continents. On the basis of their altitudinal distribution and orographic structure, highly disparate hydrological regimes can be observed over a relatively small area. They act as a fixed obstacle to the atmospheric flow, resulting in heavy precipitation on one side of the mountains and dry regions on the other. With their ability to store reserves of snow and ice at cold higher altitudes, they also play an important role in providing water supplies for surrounding low-lying regions. Through analysis of the water balance and its components, the following sections present a quantitative analysis of water resources in the Alps.

2 Precipitation

2.1 Introduction

Mountain chains such as the European Alps constitute an important factor in atmospheric circulation [1]. They trigger a variety of climatic and meteorological effects and cover a wide range of the spatial scale [2]. For example, they are manifested in the modification of the inner-continental climate zones or in the small-scale distribution of precipitation. Both can be important for the hydrology of alpine catchments.

One significant feature of mountain ranges is their barrier effects which can block or alter entire wind systems, the consequences of which can be observed not only in the mountains themselves but also much further afield. As a natural barrier, the Alps trigger convective and advective cloud formation, particularly in their peripheral areas. Hence they exhibit much more humid conditions than their adjacent environment [3]. As regards the small-scale distribution of precipitation in the mountains themselves, the differences between windward and leeward in terms of the prevailing wind directions and the effect of local wind systems are of decisive importance. Hence it is possible to determine a correlation between altitude above sea level and precipitation volumes, particularly if observed over a longer term.

2.2 The Formation of Precipitation

2.2.1 The Main Aspects of Precipitation Formation

Water is present in three aggregate states: solid, liquid and vapor. Of the estimated global water resources of 1386 million cubic kilometers, however, only 0.001% or 0.013 km^3 is stored in the atmosphere as water vapor (Table 1, [4]). If fully released, this volume of water would produce 25 mm of precipitation depth globally. Given an average annual and global precipitation of 972 mm [5], the water vapor in the atmosphere must therefore be completely replenished at least 39 times per year or approximately every nine days.

The water vapor available in the air plays a decisive part in the formation of precipitation. The amount of water that can be retained in the air in the form of vapor is primarily dependent on the temperature: Warm air masses can absorb more humidity than cold air masses, whereby there is an exponential correlation between temperature and saturation humidity (Fig. 1, [4]).

In the conditions described by the curve ("dew point temperature"), dew or clouds are formed, i.e. the water vapor condenses. If the conditions shown in the part above the curve are achieved, the condensed water vapor falls in the form of precipitation. In the conditions below the curve, water vapor enrichment or a cooling-off may occur without causing any formation of dew or precipitation. Since the air temperature is substantially determined by relief and altitude, these factors also have an impact on the maximum possible water vapor content in the air. In principle, precipitation is formed when air masses cool down, the consequences of which can be either dynamic (orographical and frontal induced precipitation) or thermal (convective induced precipitation) (Fig. 2, [6]).

Table 1 The global distribution of water [4]

		Volume	Depth
Total global water resources	100%	1,386 Mill. km^3	2,718 m
of which present in:			
Atmosphere	0.001%	0.013 Mill. km^3	0.025 m
Living creatures	$<$ 0.001%	0.001 Mill. km^3	0.002 m
Waterways and inland seas	0.013%	0.19 Mill. km^3	0.4 m
Soil water	0.001%	0.017 Mill. km^3	0.03 m
Groundwater	1.69%	23.4 Mill. km^3	45.88 m
Polar ice	1.76%	24.4 Mill. km^3	47.85 m
Fresh water, total	3.47%	48 Mill. km^3	94.18 m
Salt water seas, salt lakes	96.53%	1,338 Mill. km^3	2,642 m

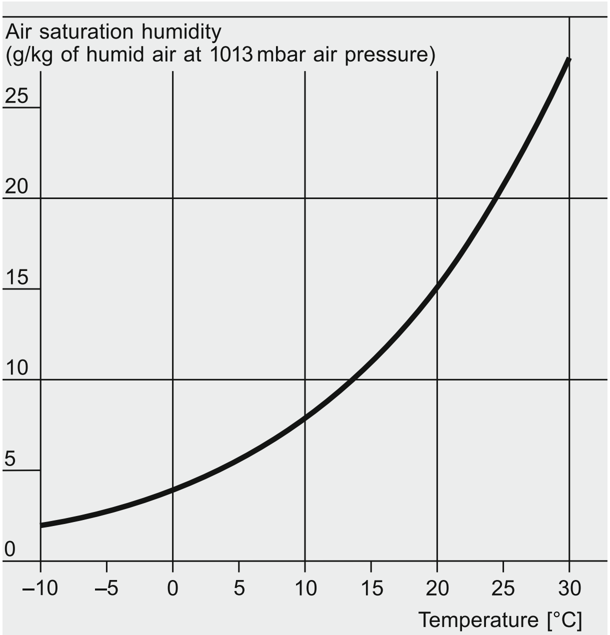


Fig. 1 Correlation between temperature and saturation vapor pressure [4]

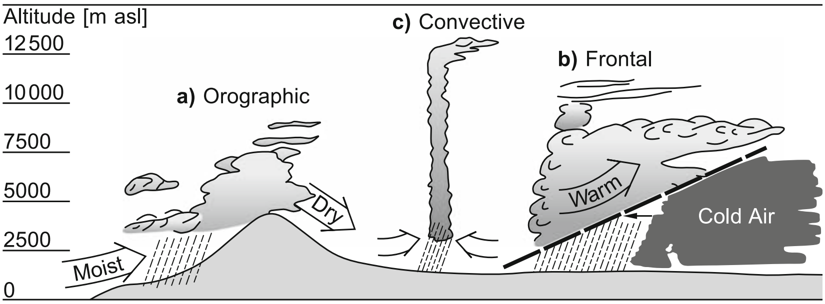


Fig. 2 Precipitation triggered by orographic, convective, and frontal processes [6]

Orographic Induced Precipitation

When humid, warm air masses are transported towards a chain of mountains by the prevailing wind systems, they are forced upwards and simultaneously cooled by the natural barrier. If, in the process, the temperature reaches or falls below dew point, clouds are formed on the windward side of the mountain range. The intensity of the precipitation triggered by this process exhibits a high spatial variability, with precipitation rates primarily dependent on the uplift distance, uplift velocity and the water vapor content in the air [7]. On the leeward side of the exposed mountain range, this situation usually gives rise to strong fall winds (such as the "Föhn"). Precipitation systems of this type are often the cause of mountain-induced dry and humid zones (Fig. 2a).

Frontal Induced Precipitation

Triggered by planetary pressure gradients, large-scale transfers of air masses occur which differ in terms of their energy or humidity content. When they collide, this creates frontal zones as the warmer, lighter (or even more humid) air rises above the colder air. This process can result in condensation and longer-lasting precipitation (Fig. 2b).

Convective Induced Precipitation

The sun warms the layer of air closest to the ground, and the heated air masses are expanded and forced upwards. As this happens, the air cools down and the water vapor retained in the air condenses. Convection flows of lengthy duration can induce cloud formations, leading to short, intensive precipitation. In the alpine region this process occurs primarily in the summer, in the form of heavy thunderstorm-induced rain, particularly if the atmospheric conditions are unstable (cold over warm, Fig. 2c; [8]).

2.2.2 The Influence of Mountains on Precipitation Formation

In addition, the processes discussed in Sect. 2.2.1 are specifically modified in mountainous regions, where the influence of the relief on atmospheric circulation can be thermal, mechanical or a combination of the two [9]. [10] has broken down these effects using the spatial scale (Fig. 3).

At this juncture, two highly typical examples of the European alpine region are discussed in more detail: On the one hand it is possible to observe on a micro scale (within the single kilometer range) the formation of local wind systems triggered by the disparate warming of the valley sides (valley-slope circulation). On the other hand, advective conditions are frequently modified at the upper meso scale (100-km range), which ultimately can result in "lee cyclogenesis" [11]. Both systems significantly affect the local and supra-regional distribution of precipitation:

Slope-Valley Circulation

Slope-valley circulation, which is exclusively dependent on thermal factors, occurs as a result of spatially differentiated insolation (Fig. 4, [12]). During the day, slopes

Influence \ Scale	Global or macro scale	Synoptic or upper meso scale	Lower meso to micro scale
Mainly mechanical	Barrier effect (wave deformation)	Windward blocking of cold air, front deformation, lee cyclogenesis	Channeling, mountain waves, fall winds
Mainly thermal	Plateau effect (large circulation systems, e.g. plateau monsoon)	Mountain-foreland circulation systems	Slope, mountain and valley winds

Fig. 3 Influences of mountains on atmospheric circulation – broken down according to spatial scale [10]

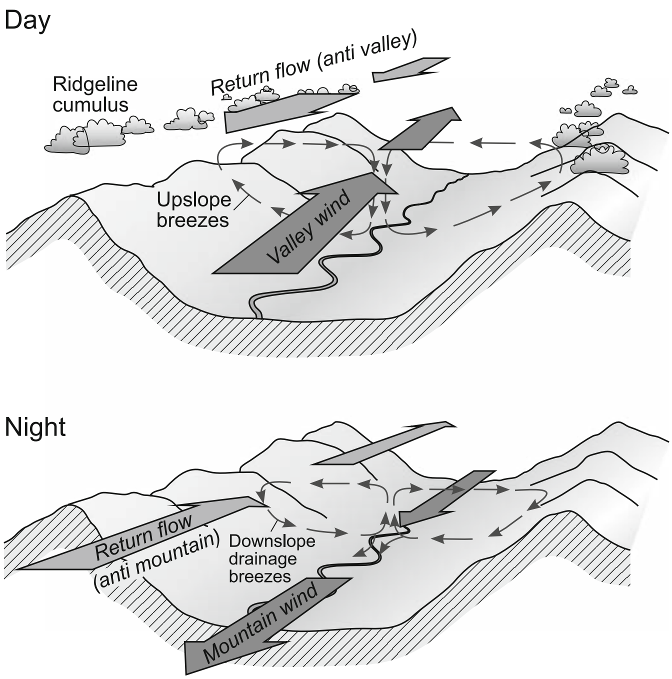


Fig. 4 Slope-valley circulation in the course of a day (vantage: valley upwards) [12]

at a higher altitude are exposed to stronger insolation than the lower-lying regions and the valley floor. The resultant warming triggers a slope upwind in all valleys, which can result in convective cloud formation and possibly precipitation in the upper slopes, particularly in the afternoons, due to the confluence of air masses. Stronger outgoing radiation during the night results in cooler air masses on the slopes, which sink towards the valley centre and outwards. This has a dissipating effect on the clouds above the mountain tops. Because of the conservation of mass balance, a vertical compensation flow is triggered over the valley centre, which can result in light cloud cover or fog above the valley floor. Because of this varying local wind system in the course of the day, valley slopes are often more humid than the valley centre.

Lee Cyclogenesis

One specific form of the mechanical effect of atmospheric circulation in the alpine region is lee cyclogenesis, which is linked to the passage of a cold front from North to South (Fig. 5, [10]). In the initial stage, the cyclones drifting in from the North trigger local southerly winds ("south Föhn"). If this process continues, the cold polar air is unable to flow over the Alps due to a lack of kinetic energy, and is forced to flow around the mountain range. This situation frequently gives rise to two divergent flowpaths which are subsequently manifested as the Mistral, Bise and Bora. In the final stage, a lee vortex is formed behind the alpine arch – typically within hours – which can strongly affect weather and precipitation volumes on the southern side of the Alps, especially in spring and autumn [10]. The approximately 30 lee cyclones which occur on average each year continue in two main directions:

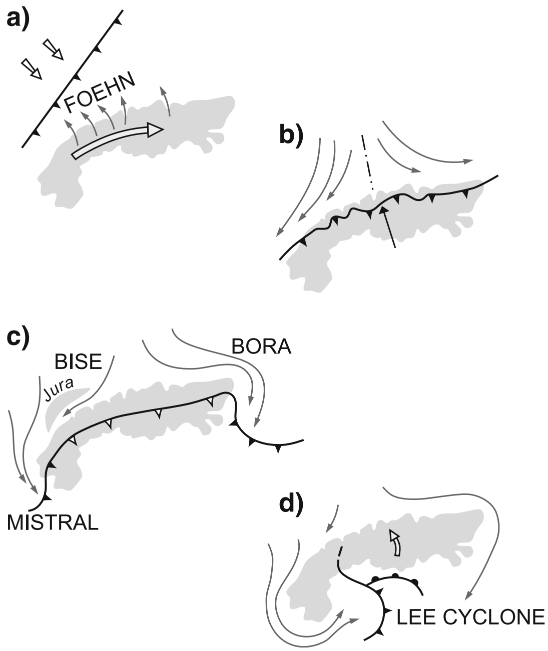


Fig. 5 Formation of a lee cyclone [10]

Either over the eastern Alps towards the interior of Eastern Europe, or along the North coast of the Mediterranean towards the Balkans [13].

2.2.3 Altitude Dependency of Precipitation

The dependency of precipitation on altitude was the subject of numerous studies in the twentieth century, the findings of which were widely disparate (e.g. [5, 14–20]).

In theory, since the absolute water vapor content of air reduces as the temperature sinks or the altitude rises, the volume of precipitation should decline. Yet the opposite is the case, particularly in mountainous regions in the temperate latitudes: here the annual volumes of precipitation generally tend to rise with the altitude. This is the result, on the one hand, of higher wind speeds at higher altitudes, which cause a relatively large shift in humid air masses. On the other hand, precipitation occurs more frequently and often at a much greater intensity.

Today the assumption is that there is no direct causal correlation between altitude and precipitation volumes. Rather, the influence exerted by the relief on approaching air masses is of decisive importance for precipitation volumes [19]. These effects also vary depending on weather conditions, climate region and season. Moreover, the correlation between precipitation volumes and altitude becomes significantly weaker the shorter the duration of the event ([21], cf. also Fig. 10).

The data available on altitude zones in the Alps with the highest average annual precipitation also varies widely (e.g. [16, 18, 19, 22]). Although this area cannot be precisely determined due to the lack of measurements or due to unreliable statistics, and despite the fact that major spatial differences have to be taken into account, current studies indicate an altitude between 3,000 and 3,500 m ASL [20].

2.2.4 Forms of Precipitation

Precipitation can occur in solid or liquid form. There are also differences in the size of particles or droplets, and hence in fall velocity [1]:

Solid: Snow, hail, ice, snow pellets, frost pellets, hoar frost, rime

Liquid: Rain, drizzle, fog deposition

In addition to liquid precipitation, snow in particular is an important contributor to the water balance in alpine catchments, since it acts as a temporary storage depot. Snow is formed in clouds as a result of condensation at temperatures below zero degrees Celsius. The aggregation of humidity at solid (ice) particles creates large crystals which fall to the ground in the form of snowflakes given the right conditions. Snow falls only at low speed, and is therefore usually carried by the wind over wide areas away from its atmospheric place of origin.

However, fog can also contribute to precipitation. This so-called fog deposition occurs as a result of drifting fog and clouds, whereby the volume of precipitation is determined by the drift velocity, density of droplets and the attributes of the vegetation which “filters” the water from the air. Thus, the most common abundant fog deposition can be expected in alpine forests, at high altitudes, on ridges and peaks, and is probably the most common source of water at altitude zones between 2,000 and 3,000 m ASL. Fog deposition often occurs as transient precipitation, i.e. a substantial portion is re-evaporated.

Locally, fog deposition can account for a significant proportion of the total volume of precipitation (e.g. [23, 24]). Yet a high local variability in fog deposition characteristics must be assumed, particularly in the alpine region. Hence the water balance in larger alpine catchments is only slightly distorted if fog deposition is not taken into consideration [5].

2.2.5 Measuring Precipitation

In principle, capturing and measuring precipitation is a simple matter. But its measurement is strongly influenced by the wind field prevailing at the measuring device and in its vicinity. In addition, losses due to splashing, evaporation from the device and inaccuracies in readings affect the quality of the measurement. In winter the loss of captured precipitation increases as the proportion of snow increases or due to its low fall velocity and its wind drift. This is the reason for the relatively large systematic precipitation measurement errors exhibited by mountainous regions: Errors of up to 15% have been recorded for higher altitudes and more than 50% if the proportion of snow is large [25].

Another problem encountered when registering precipitation in mountainous regions is the spatial distribution of measurement stations (e.g. [26]). The volume and intensity of precipitation in mountainous regions is highly variable even over small areas. Yet these very regions often suffer from a lack of dense precipitation measurement networks, since measurement stations are mainly to be found in valleys. Consequently, up to now the data collected in order to measure the spatial variability and volume of precipitation in higher-altitude regions has been insufficient.

2.3 Precipitation in the European Alpine Region

On an international scale, the Alps are a middle-sized chain of mountains which, due to their situation in the central latitude of Europe, are influenced by maritime as well as continental factors. Humidity is generally transported by the west and south winds flowing from the Atlantic or the Mediterranean towards the mountain chain. With altitudes of up to 4,500 m ASL, the Alps present an enormous barrier to the air masses being transported in this way, and this barrier effect reinforces European meridional temperature gradients [10].

2.3.1 General Characteristics of Precipitation

The spatial distribution of average annual precipitation for the period 1971–1990 (Fig. 6, [27]) clearly illustrates the significant role that the Alps play in the regional distribution of precipitation: The volume of precipitation already starts to rise in the lower regions adjacent to the Alps compared to regions with the same altitude above sea level further afield [28].

The two vertical profiles through the Swiss Alps on the precipitation map (Fig. 7, [29]) also illustrate the correlation between annual precipitation and altitude, particularly in the peripheral areas of the northern alpine region, and to some extent also the southern alpine region. Towards the main alpine ridge, however, average precipitation volumes decline again despite the increase in altitude [27]; in specific inner-alpine regions, even lower precipitation volumes than along the alpine periphery have been measured. The shielding effect of the mountains is particularly evident in the dry valleys, which are influenced by flanking lee effects (Switzerland: Valais, Engadine; Italy: Vintschgau).

The large-scale multi-year precipitation patterns for the European alpine region as shown in Figs. 6 and 7 can be summarized as follows [5]:

- Precipitation is greater in the outermost areas of the alpine arc than in the inner-alpine areas; moreover, precipitation declines in the western Alps towards the

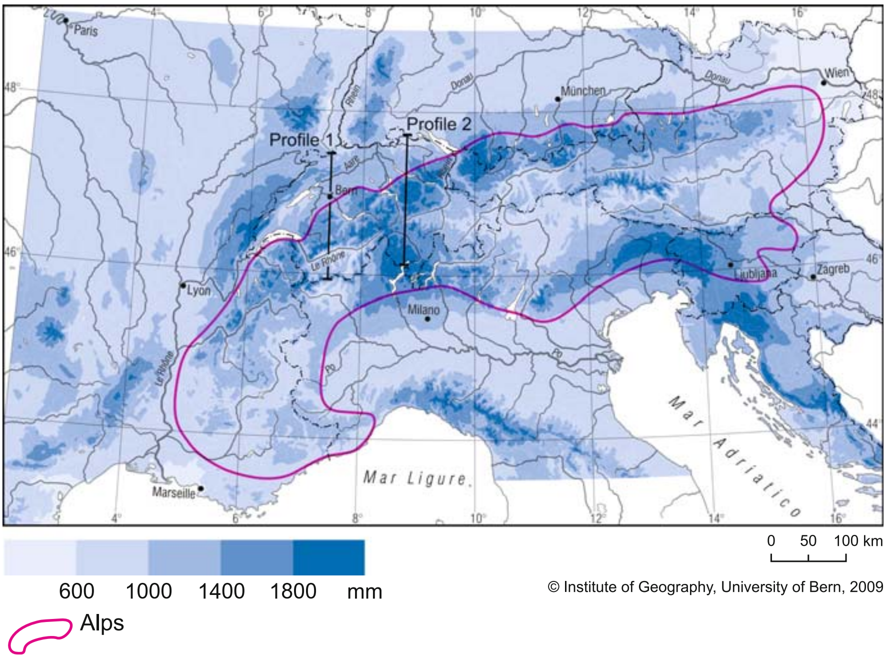


Fig. 6 Distribution of average annual precipitation in the European alpine region (1971–1990) [27]. Profiles 1 and 2 see Fig. 7

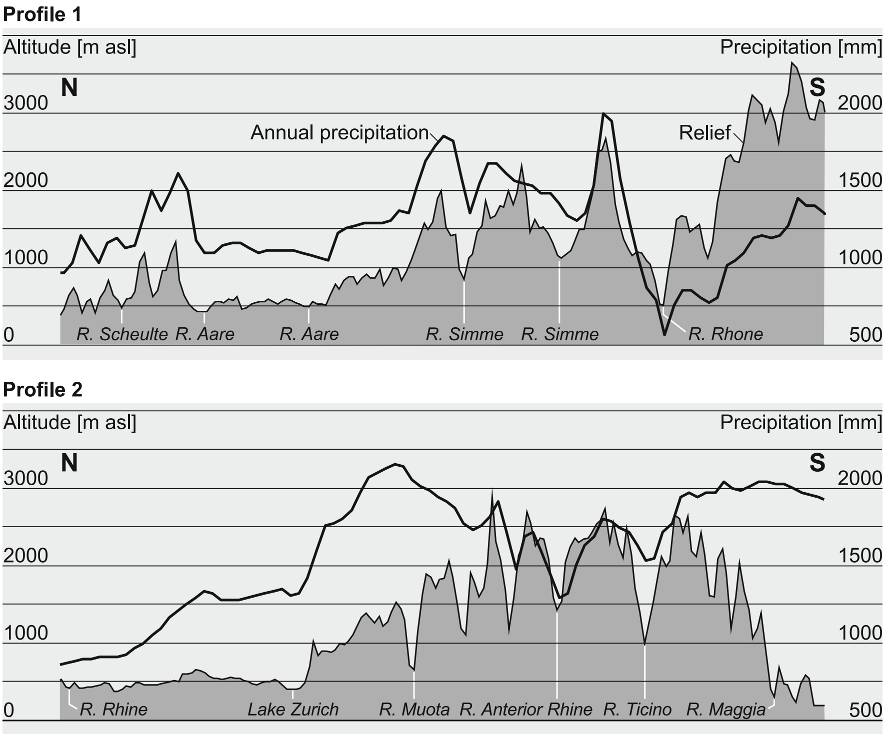


Fig. 7 Precipitation profiles through the Swiss Alps (average annual precipitation totals, 1971–1990), data: [27], [29]. Location of profiles see Fig. 6

south-west and in the central Eastern Alps towards the east. The southern periphery of the Alps has regions that exhibit particularly high precipitation volumes or maximum volumes for the Alps.

- The low-precipitation zone of the central Alps covers a larger area in the east than in the west, and is interrupted in the Gotthard massif by a region of maximum precipitation. The reason for the varying spatial distribution of dry zones is that narrow valleys predominate in the west, while the relief in the east is less pronounced; producing larger connected regions with low precipitation.
- While Figs. 6 and 7 show the correlation between orography and annual precipitation, this correlation is far from constant over larger areas and is subject to major local variations.

The following spatial patterns are evident from the seasonal distribution of precipitation (Fig. 8) [30]:

- Winter (e.g. January): In contrast to other seasons, precipitation over the entire alpine region is relatively low, and the central alpine regions are particularly dry. The low mountain ranges on the western side of the northern Alps

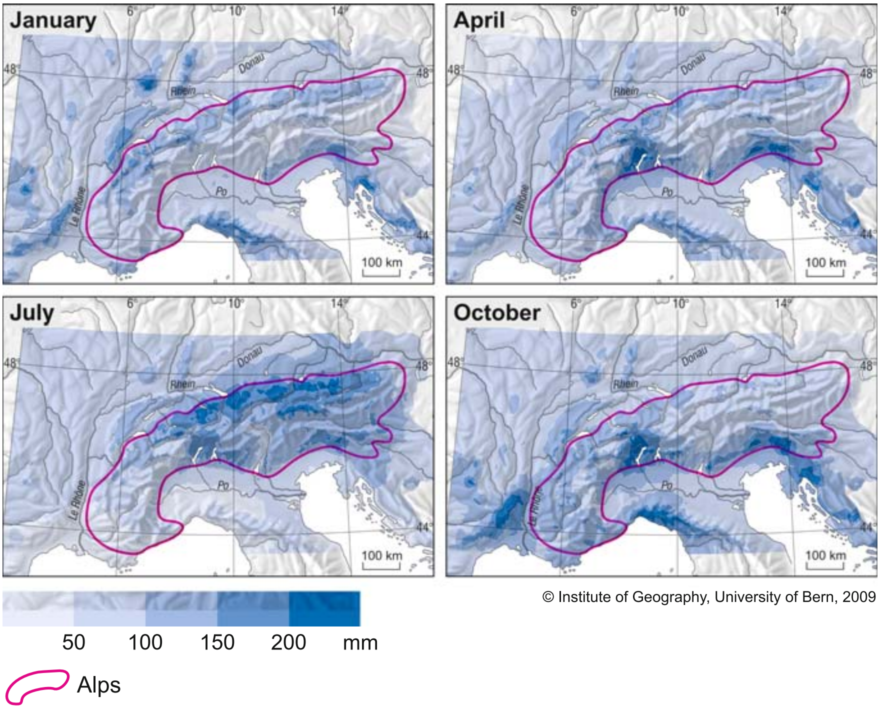


Fig. 8 Distribution of selected average monthly precipitation in the European alpine region (1971–1990) [3]

(Jura, Black Forest, Vosges), however, receive more precipitation than during other seasons.

- Spring (e.g. April): Compared to winter, the distribution of precipitation on the northern side of the Alps is much more balanced during this season. Particularly high volumes of precipitation occur along the southern periphery of the Alps.
- Summer (e.g. July): The southern lowlands (Po plain) exhibit extremely dry conditions, whereas high volumes of precipitation are recorded in the central and eastern Alps. The greatest volume of precipitation falls along the northern periphery of the Alps; precipitation in the northern lowlands is accordingly higher than at other times of the year. Precipitation is triggered primarily by storms.
- Autumn (e.g. October): While the distribution of precipitation is similar to that exhibited in spring, the Central Massif as well as the south-eastern periphery of the Alps exhibit particularly high precipitation. Because of more intensive evaporation of warm maritime surfaces, coupled with greater frequency of low-pressure zones above the Mediterranean at this time of the year, more

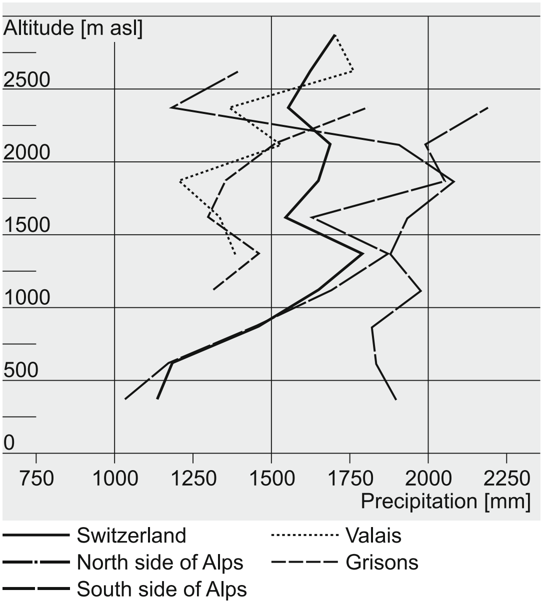


Fig. 9 Average annual precipitation in various regions and altitude zones in Switzerland (1961–1990) [31]

humidity is transported in the direction of the Alps, often manifesting itself in the form of heavy precipitation along the southern Alps.

The spatially variable influence of topography on average annual precipitation in the alpine region is also reflected in the altitude gradients of various climate regions of Switzerland (Fig. 9). In the study conducted by [31], the average annual regional precipitation was calculated as a remainder of the water balance. In general, the altitude dependency of precipitation occurs primarily in regions below 1,500 m ASL, although the correlation is not strong and can vary from one region to another. For regions above 1,500 m ASL, in most cases the relationship exhibits no linear progression; at times the average annual precipitation volume has even been observed to fall as the altitude increases. Moreover, the central alpine dry zones – illustrated here by the example of the Valais – are clearly evident. Furthermore, the precipitation gradient for the south side of the Alps is smaller compared to the northern side of the Alps, although overall it exhibits higher levels of precipitation.

The differences in precipitation characteristics to be observed on the north and south side of the Swiss Alps are largely a result of the different proportions of advective and convective precipitation events, as well as the differing intensity of precipitation. However, as mentioned above, topographical characteristics can also exert a significant influence [32]. Precipitation maps for the European alpine region (Figs. 6 and 8) have taken this fact into account by calculating around 10,000 local gradients in order to map precipitation altitudes.

2.3.2 Heavy Precipitation in the European Alpine Region

Heavy precipitation is defined as precipitation events of major intensity which therefore occur relatively seldom. A threshold value per station or climate region, or an exceeding probability, is frequently applied to determine heavy precipitation events [21].

The general principle is that the intensity of a heavy precipitation event weakens the longer it lasts. Observations in the Swiss alpine region show that the intensity of precipitation is slightly higher on the north side of the Alps than on the south, particularly for short-duration events. Conversely, the maximum heavy precipitation of long duration ($\geq$1 day), which is primarily triggered by advective conditions, is greater in intensity on the south side than on the north side of the Alps. The most intense precipitation occurs on the northern and southern peripheries of the Alps [29].

The pattern of altitude dependency in the case of heavy precipitation in the alpine region differs clearly from the average or monthly precipitation volumes. A comparison of transversal profiles through the Swiss Alps (Fig. 10, [21]), which include the extreme precipitation (here with a 100-year recurrence period) occurring in various observation periods ranging from 1 h to 1 year, as well as the average annual precipitation between 1901 and 1970, enables the following patterns to be determined: The altitude dependency, which is still clearly evident for average annual precipitation volumes, becomes weaker for heavy rainfall events with shorter observation periods, and virtually disappears in the range of hourly volumes. Within this time scale, topographic conditions in the close vicinity or further afield influence the precipitation process more strongly than is the case over longer observation periods [21, 33].

3 Runoff

3.1 Introduction

Runoff is a core element of the water balance, and as such has a reciprocal relationship with the community. On the one hand, river water – often referred to as “blue water” [34] – constitutes an important resource. For instance, it is essential for irrigation farming, which accounts for around 20% of the world’s food production. On the other hand, when it takes the form of high water or flooding, it presents a danger to humans and their infrastructure. High water and flooding account for around 30% of economic losses [35], and are ranked among the world’s most damaging natural disasters alongside earthquakes and storms. Both aspects—resource and natural hazard—are accorded particular importance in the alpine context.

Information on runoff characteristics in alpine regions provides the key to a comprehensive understanding of alpine hydrology. Runoff is both an expression of the complex interplay between precipitation, evaporation and storage changes, and of the close relationship with natural environmental conditions.

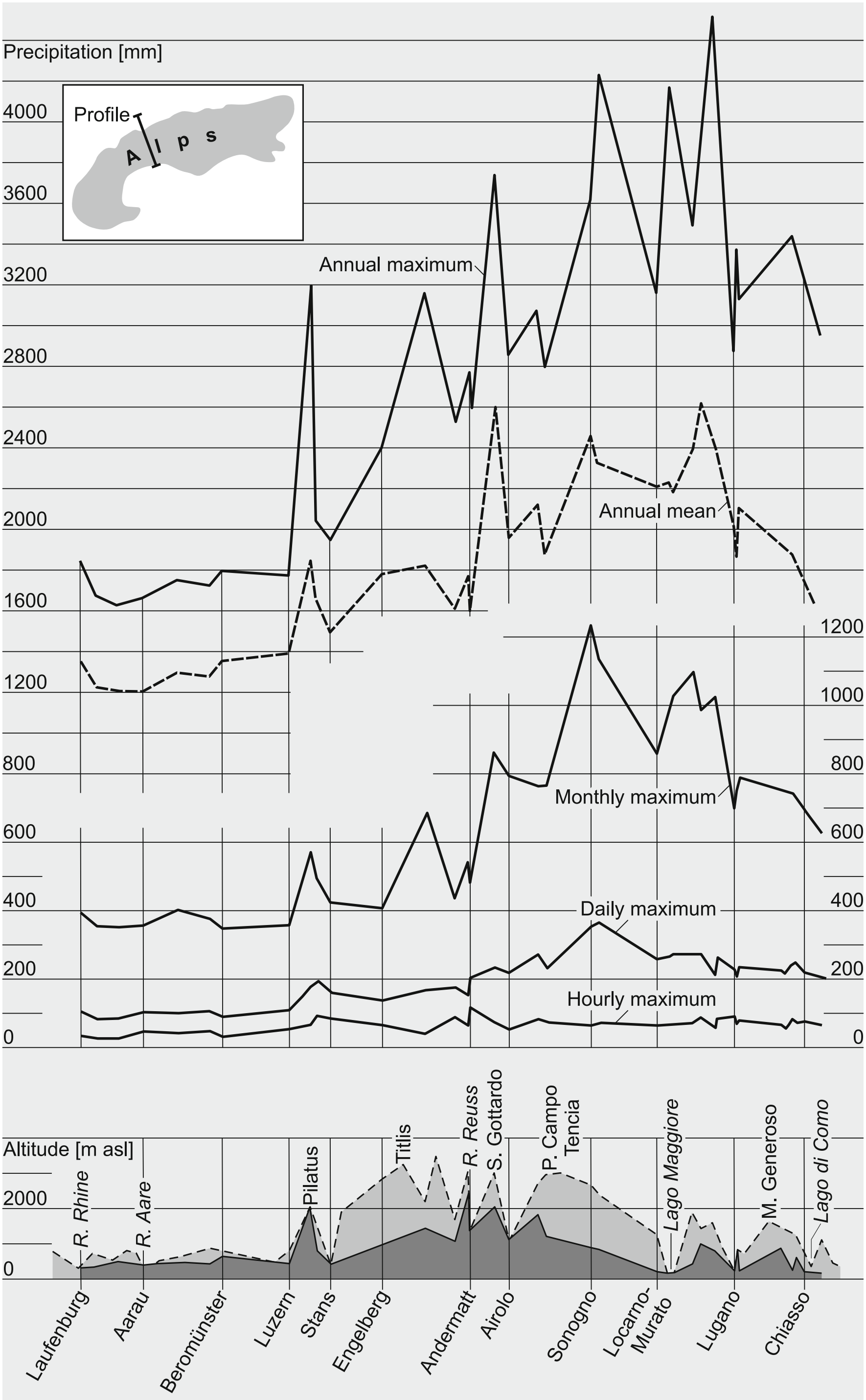


Fig. 10 Precipitation profile through the Swiss Alps (100-year heavy precipitation observed over periods of varying duration, and average total annual precipitation; 1901–1970 [21]

3.1.1 Measurement and Measurement Networks

Runoff is generally measured in two steps: (1) (continuous) recording of the water level, and (2) conversion of the water level to discharge using the rating curve, which relates the vertical depth of water in the stream to flow (volume per unit of time). To obtain the rating curve, it is necessary to determine the runoff discharged from different water levels directly. The hydrometric current meter is often used for this purpose, although more recently this has been replaced in larger rivers by the Acoustic Doppler Current Profiler (ADCP). The dilution method, which uses salt or fluorescence tracers, is ideal for smaller, turbulent alpine rivers and streams [29].

Runoff measurement stations are often expensive to install and operate in alpine regions. For example, the data collection points must be specially protected against derogation by solid matter, and streambed erosion as well as bed aggradation poses a major problem. Measurement of runoff in mountain torrents is particularly difficult and costly [36]. In general, the high cost of measurement and the inaccessibility of many data collection sites are the main reasons for the comparatively low density of measurement networks in alpine regions. Yet it is in these very regions, with their complex natural conditions, that a dense measurement network is essential in order to obtain a comprehensive overview of the hydrological conditions. In this context [37] states that it is a paradox that there is insufficient measured data on key hydrological areas. Runoff measurement networks in the Alps [38, 39], however, do not quite fit this picture. In fact, the volume of available data here is relatively good, although runoff at a great many stations is heavily influenced by hydropower production [40], see also Sect. 3.2.2

Fig. 11 illustrates this “hydrological paradox” [37] by comparing the altitude distribution of runoff measurement stations in Switzerland and Nepal: Nepal, a key hydrological mountainous country for the Indian subcontinent (cf. Sect. 5), has only

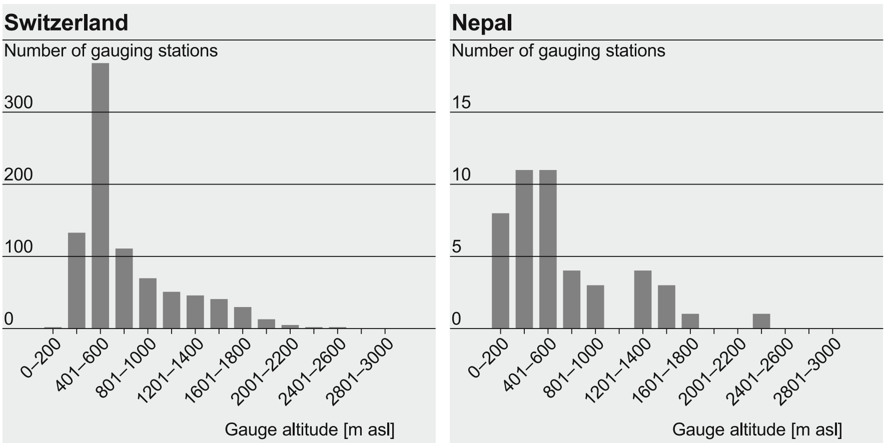


Fig. 11 Frequency of runoff measurement stations per altitude level (status 2005), *Left* = Switzerland, *Right* = Nepal (based on [38, 80])

a few runoff measurement stations. Indeed, the latter are all located at a lower altitude than data collection stations in Switzerland, despite the pronounced difference between the orography of Nepal and that of Switzerland. The Nepal example clearly illustrates what [41] is referring to when he complains that the mountains represent "the blackest of black boxes in the hydrological cycle."

3.1.2 Basin Characteristics

It is apparent from the above that only very few catchment areas in alpine regions have access to runoff data. Consequently, regionalization procedures need to be developed and applied for the purpose of estimating runoff characteristics [42]. In turn, however, the value of such estimates depends on the runoff data available for calibrating these models. This gives rise to the same dilemma mentioned above in connection with runoff measurement networks: In alpine regions with few runoff measurement stations it will be difficult to develop effective regionalization approaches.

As a rule, regionalization approaches are based on procedures to define the parameters of the catchment areas using basin characteristics, and modeling the relationships between these characteristics and the hydrological characteristics of interest [42]. Basin characteristics enable the numerical representation of climate conditions, topographical, pedological, and geological characteristics, land use and other aspects which impact runoff characteristics. Many of these key indicators are dependent on the altitude, giving rise in some cases to strong correlations between the hydrologically relevant key indicators. This must be taken into account when developing stochastic models. Fig. 12 shows the change in various basin parameters

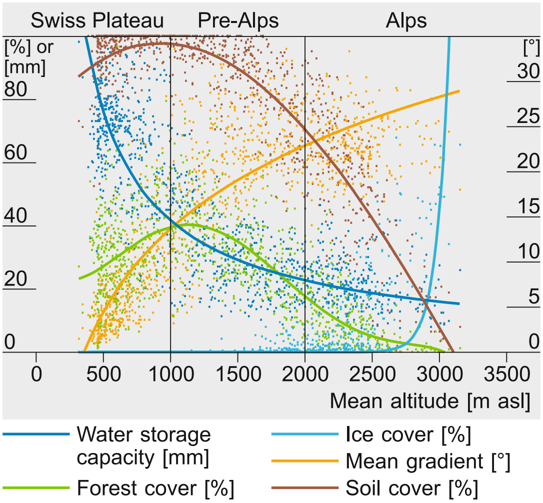


Fig. 12 Change in various hydrologically relevant basin characteristics with increasing mean altitude [81]. Points: Data from the 1,068 base regions in Switzerland. Line: Average change in key indicators with altitude

with altitude. This relationship can be used as the basis of fundamental considerations on the behavior of important basin characteristics in alpine regions:

The water storage capability serves as an indicator for the total water storage capacity of the soil and is based on the water available to the plant in the root zone. Fig. 12 shows how the soil's storage capacity drops off sharply the higher the altitude.

In Switzerland the greatest forest cover is located in the pre-alpine regions, i.e. between 1,000 and 1,500 m ASL. Forest cover has a significant impact on interception: [43] estimated, due to interception by forests in the Eastern Alps, 20–30% of annual precipitation does not reach the soil and is hence withheld from runoff. Moreover, the density of roots in the forest soil increases the stability of the slopes, which must be regarded as beneficial in terms of protection against natural hazards. In the case of flooding or high water, the impact of forests is a matter of dispute. [44] succinctly sums up the debate on the forest hydrological hypothesis as follows: "Forests presumably reduce flood flow and increase base flow. The debate about this forest hydrological hypothesis still goes on at the system scale of catchments. The investigations of [45] as well as newer aspects in soil hydrology at the process scale may help to differentiate: forest soils are more likely to control run-off than soils under any other land-use. However, not all the soils under forest bear the characteristics which support the forest hydrological hypothesis, and soils under various vegetation covers may show well developed properties to effectively mitigate peak run-off."

Soil cover declines sharply above 1,000 m ASL, giving way to increasingly large areas of debris and rock cover. Because of the high proportion of debris and rock cover and the soil's low water storage capacity, faster water turnover, higher runoff coefficients and, in general, extremely high volumes of runoff can be expected (see Sect. 3.3).

In Switzerland's alpine region, glaciers occur above around 2,500 m. This has a major impact, on seasonal flow regimes, among other things (see Sect. 3.2).

3.1.3 Structure of the Section on Runoff

The following discusses relevant aspects of flow characteristics in alpine regions, using the example of the European alpine region which in terms of data is well documented, and predicated on the classical mean water – high water – low water breakdown.

3.2 Mean Water

3.2.1 Mean Annual Runoff

Depth of runoff [mm a^{-1}] can be used to obtain a direct comparison of the mean annual flow rates from drainage basins or geographical regions of different sizes.

Table 2 Characteristics of runoff depth in regions and river basins in Switzerland (basis of data: [51])

Region	Runoff [mm a^{-1}] at 1,500 m asl.	Gradient [mm m^{-1}]	Coeff. of determination
Northern part of the Alps	1,166	0.55	0.51
Rhine	1,018	0.41	0.48
Aare (without Reuss and Limmat)	1,193	0.71	0.79
Reuss and Limmat	1,443	0.58	0.42
Inner alpine zone:			
Rhone (without Jura)	856	0.35	0.26
Inn (mean altitude $\geq$1500 m asl.)	(300)	0.70	0.3
Southern part of the Alps:			
Ticino	1,469	0.29	0.34
Switzerland	1,073	0.37	0.32

Table 2 provides an overview of the spatial variability of runoff depths in the Alps. A simple linear regression $D = f(\text{mA})$ (where D = Depth of runoff and mA = mean altitude) was derived for the regions shown in the table, on the basis of around 200 mesoscale catchments in Switzerland [46]. The significance of the correlation between altitude and runoff depth is derived from the coefficient of determination, providing a good to very good explanation of the spatial variation in runoff on the north side of the Alps. Here the runoff increases by an average of 0.4–0.75 mm m^{-1}. This behavior correlates with the distribution of precipitation, which is also altitude-dependent. Map 5.7 of the Hydrological Atlas of Austria shows that regional variations in runoff depth in the Eastern Alps are more pronounced than variations in precipitation, since "the distributions of precipitation and evaporation are generally contrary" [47]. According to studies conducted by [48], annual precipitation on the north side of the Alps is increasing by approximately 0.7 mm m^{-1} (cf. Sect. 2.2.3), and according to analyses performed by [49], annual evaporation is declining by 0.22 mm m^{-1} (cf. Sect. 4.4.2). As with precipitation, the role played by mean altitude in explaining the spatial variability in runoff in inner-alpine regions and south of the Alps is nonexistent or, at best, secondary.

According to [50], the lower border of the alpine-influenced catchments can be located (in Switzerland) at a mean altitude of around 1,500 m ASL. Above this altitude, flow characteristics are dictated by snow and glaciers (see below). Hence a comparison of runoff depths at the lower edge of the alpine region is shown in the second column of Table 2: the precipitation-related higher runoff on the south side of the Alps and the relative scarcity of runoff in inner-alpine zones is pronounced. Because of the varying gradients between the north and south side of the Alps, similar runoff depths can be assumed from mean altitudes of 2,500 m.

The influence of the mean altitude on mean runoff conditions can also be seen in a comparison of the two alpine states of Switzerland and Austria: Switzerland's mean runoff depth is around 1,000 mm a^{-1} [51] at an average altitude of 1,300 m ASL, while Austria's is 630 mm a^{-1} [47] at an average altitude of 770 m ASL.

3.2.2 Seasonal Runoff Characteristics

Seasonal runoff distribution can be used to determine the complex process of runoff generation. Regions with similar seasonal flow characteristics can be categorized by type. The best-known regime classification is from [52]. The criteria he used were (1) the dominant process of runoff origination (e.g. "glacial" i.e. fed by glaciers), (2) the number of maxima and minima for mean monthly discharges, and (3) the monthly dimensionless Pardé coefficients. The latter are calculated based on the quotients between the mean monthly and mean annual discharge. An overview of methods for classifying runoff regimes is provided by [50] and [53].

Seasonal fluctuations in runoff in alpine catchment areas are influenced by snow accumulation in early winter, the build-up of snow cover in winter, snowmelt processes in spring and early summer and – where applicable – glacier melt in summer (Fig. 13). The dominant role played by snow and ice in seasonal discharge volumes gives rise to single-peak regimes. Multiple-peak regimes, on the other hand, arise due to the alternating influence of precipitation, snow accumulation, snowmelt and evaporation. They typify northern alpine catchment at lower altitudes, as well as most southern alpine catchment areas which are influenced by the Mediterranean climate.

Because of the varying impact of snow and glaciers, six regime types have been classified for Switzerland [50]. These types differ primarily in terms of the flow characteristics during the months of May to September, i.e. the period during which ice and snow are melting (cf. Fig. 14). Discharges in May and June are largely dictated by snowmelt, while those in the July to September period are influenced by glacier melt. The highest monthly discharges occur in July and August in glacial regimes, and in May and June for nival regimes. The different types of regime are distinguished on the basis of the sequence of mean monthly discharges in the months May to September. In the winter half-year, minimum discharges with

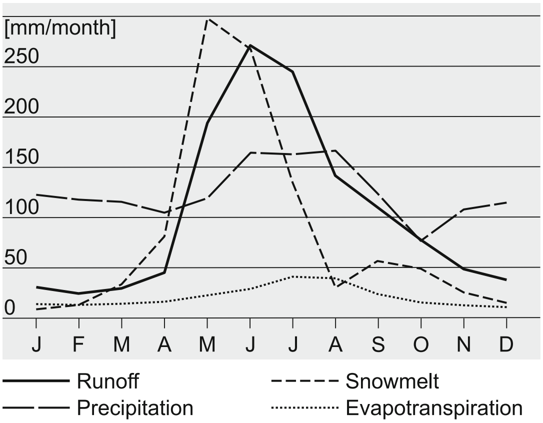
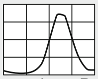
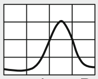
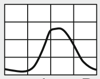
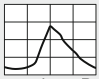
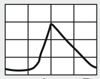
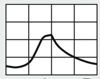


Fig. 13 Average seasonal course (1982–2000) of the main water balance elements in the alpine Dischma catchment simulated with PREVAH and the EMA snowmelt module. P: areal precipitation, R: runoff, ET: evapotranspiration and SN_M: snowmelt [82]

Regime	a-glaciaire	b-glaciaire	a-glacio-nival	b-glacio-nival	nivo-glaciaire	nival
	 Jan Jun Dec	 Jan Jun Dec	 Jan Jun Dec	 Jan Jun Dec	 Jan Jun Dec	 Jan Jun Dec
Rank	1. July 2. August 3. June 4. September	1. July 2. August 3. June 4. September	1. July 2. June 3. August 4. May	1. June 2. July 3. August 4. May	1. June 2. July 3. May 4. August	1. June 2. May 3. July 4. August
Cv (Jun)	21	21	16	17	16	20
Cv (Jul)	11	13	14	21	19	24
Basin characteristics	mA >2400 Gla ≥36	mA >2100 Gla 22–35.9	mA >2400 Gla 12–21.9	mA 1900–2300 Gla 6–11.9	mA 1550–1900 Gla 3–5.9	mA 1550–1900 Gla 0–2.9

Rank: Based on mean monthly flow [m^3/s] mA: Mean altitude [m asl]
Cv: Coefficient of variation [%] Gla: Glaciation ratio [%]

Fig. 14 Alpine regimes in Switzerland [83]. In their regime classification for Austria, [53] also distinguish between six single-peak regime types; however, their classification method differs from the one selected for regimes in Switzerland

Pardé coefficients <0.5 occur due to the storage of precipitation in the form of snow and ice. The seasonality of precipitation exerts only a minor influence on seasonal flow characteristics, since the greater volume of precipitation north of the Alps in the summer half-year is offset by higher evaporation during the summer.

Since snow and glaciers are particularly sensitive to any rise in temperature, climate change significantly influences seasonal flow characteristics [54, 88].

Not only does the dominant influence of snow and glaciers result in a single-peak regime; it also results in relatively low interannual runoff variability [55]. Figure 15 summarizes the classified Pardé coefficients for the four Swiss catchment areas over the observation period 1993–2006. The "colored carpets" enable the variability from year to year as well as the representativity of the mean seasonal course for a single year to be determined. In the alpine catchment area of the Rhône, the change in seasonal pattern from one year to another is minimal; the Pardé coefficients derived from monthly means over several years are therefore representative for any individual year. This regularity is less pronounced in catchment areas outside the alpine zone [56]. In terms of water management, these findings mean that even short-term measurements can provide meaningful information on seasonal flow characteristics in the alpine zone [57].

High discharge volumes, low runoff variability, and high relief energy provide the basis for major hydropower potential in the Alps. Around 70 TWh of electricity a year is generated from hydropower in Austria and Switzerland together. This corresponds to 50–60% of Switzerland's total electricity production [58] and 60–75% of Austria's (www.energyagency.at), depending on weather conditions and the demand structure. In some cases this use of hydropower causes massive changes in (seasonal) flow characteristics in the underlying water network. [40] and [42] have mapped and quantified the extent of this impact.

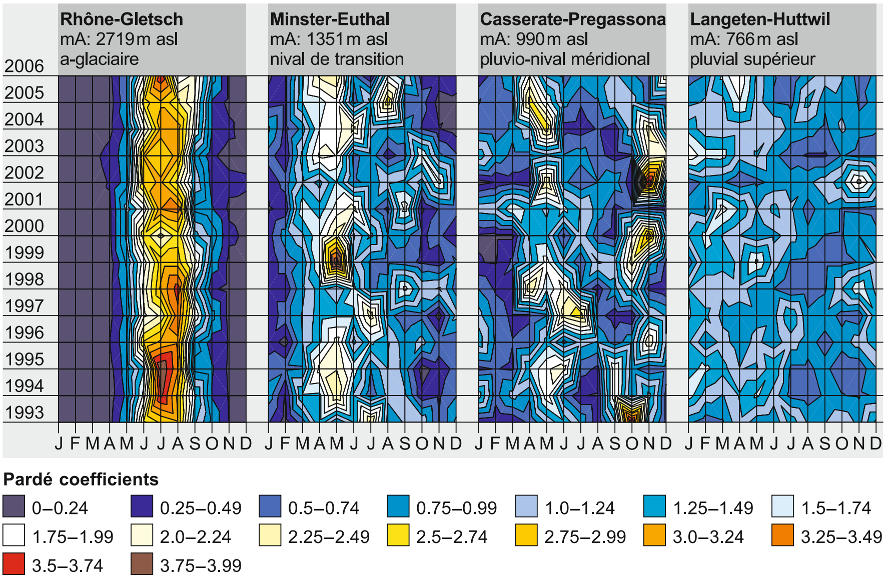


Fig. 15 Interannual variability of runoff regimes from 1993 to 2006, based on classified Pardé coefficients of selected catchment areas at varying mean altitudes [56]

3.3 Floods

Observation of flood disposition in alpine catchment areas – in other words the disposition of the region for high water [59] – provides a good starting point to the topic. [42] proposed a method for assessing the flood disposition of catchment using the quotients between the areas contributing to the flood and the total surface area of the region in question. The contributing areas are determined using an approach suggested by [60], by taking into account the density of the channel network and the topography. The runoff disposition of a catchment increases as the proportion of contributing areas rises, as has been illustrated by a comparison with the runoff coefficients of the most extensive floods measured in Switzerland. An alternative, descriptive method used to describe flood disposition is predicated on the fact that this disposition is decisively influenced by a few hydrologically relevant basin characteristics (Fig. 16).

With reference to the European Alps it can be said that there is a far greater likelihood of intense floods with high specific discharges in catchments above an altitude of 1,000 m. The upper limit of this critical zone lies at around 1,800 m, although it can vary considerably according to microclimatic and topographical conditions. Above 1,800 m, in most of the cases shortterm snow storage of precipitation reduces the hazard of floods. In other mountain areas these altitudinal limits vary according to general climatic conditions. However, the links between the elements which govern flood disposition mentioned here apply as a rule to other mountain areas, too.

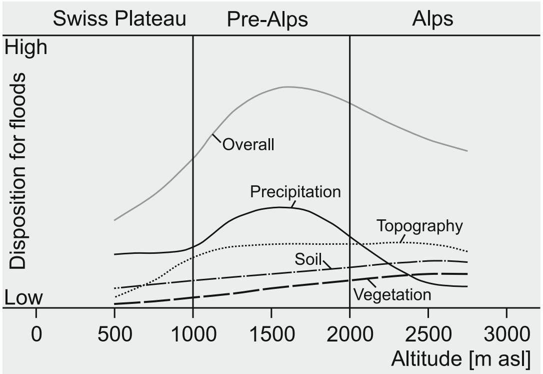


Fig. 16 Flood disposition versus altitude [84]

Quick responses are typical for the most vulnerable zone between 1,000 and 1,800 m. They are produced by high rainfall intensity in combination with steep gradients and thin soils. In many cases an extensive network of streams ensures a high specific discharge. The processes of bedload mobilization and transport are stimulated by overland flow, which is an important component of runoff generation in this zone.

High runoff disposition in this zone is furthermore evident

- in the ratio between the 100-year and mean annual flood discharge: In this zone the 100-year discharge is only around two to two and a half times as much as the mean annual flood discharge [61];
- in the high runoff coefficient around 0.5 (median) that is based on the fact that the frequent precipitation, which often takes the form of convective rainfall in summer, may fall on soil which has only a limited capacity for storing water [61].

With regard to the last point, [62] rightly point out that the storage capacity of the soil and subsoil is not exhausted everywhere, even in the case of extreme precipitation: "It is therefore important to determine the limit beyond which a catchment is virtually incapable of storing any more water." However, steep mountainous catchments are only capable of storing low volumes of water and generally react rapidly. The authors [62] then use case studies to demonstrate that in many alpine catchments the response can also be slower, which ultimately results in major spatial variability in the flood characteristics of alpine catchments.

[63] for Austria and, on this basis, [64] for Switzerland, have analyzed and classified the occurrence of large floods in the alpine zone. Using various decision criteria as a basis, they obtained five event types: Flash floods, short-rain floods, long-rain floods, rain-on-snow-events and snow- (glacier-) melt events, which are shown in Fig. 17 for selected catchments in Switzerland. On the north-facing slopes of the Alps there is one clearly identifiable zone which is dominated by flash floods and short-rain floods. This zone is largely identical with the aforementioned zone of increased flood disposition. In the alpine zone itself, which, due to the use of

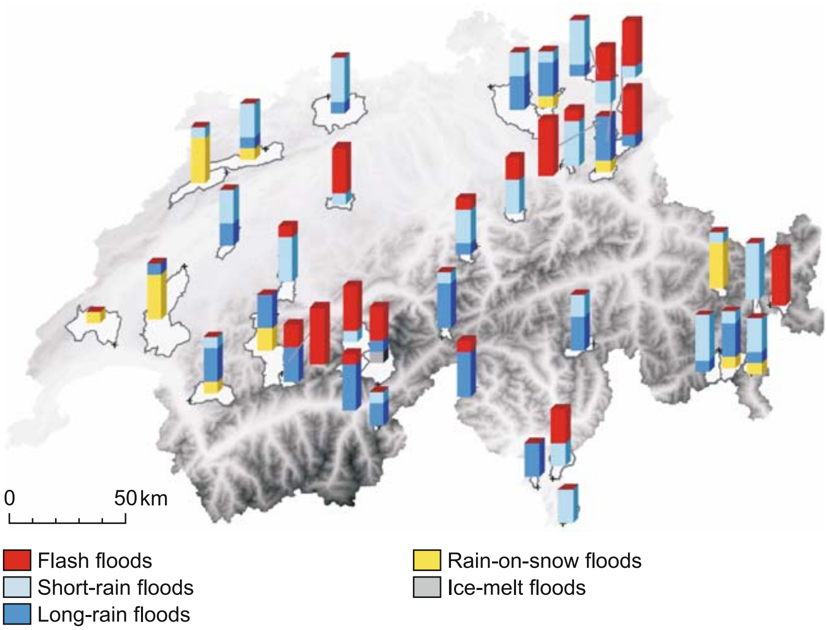


Fig. 17 Weighted frequency of the various process types [64]. Only flood events with a recurrence interval of more than 10 years are considered. These events are weighted as follows: A process type which results in an event with a recurrence more than 30 years is accorded the weighting 3. A recurrence of more than 20 but less than 30 years corresponds to a weighting of 2, and a recurrence of more than 10 but less 20 years corresponds to a weighting of 1. The weightings are then aggregated for each process type per catchment

hydropower, currently has only a few measurement stations with natural discharges (see above), the difference in frequency between the various process types is striking, and hence also the spatial variety of flood events. Long-rain floods are more relevant in the central and western regions of the Swiss Alps than in the eastern region, where short-rain floods are very frequent. Also notable is the fact that rain-on-snow events in the alpine zone are relatively rare; this is because flood peaks are more frequent in high summer, when the majority of catchments are free of snow and the zero degree line for the triggering precipitation event is relatively high, so that large areas are rained on and hence contribute directly to the runoff (Fig. 18). The opposite occurs in catchments outside the Alps, where the highest annual flood is spread evenly throughout the year.

3.4 Low Water

Low water refers to water levels which are below the limit defined as normal. As with high water, low water usually occurs at a specific time of the year depending

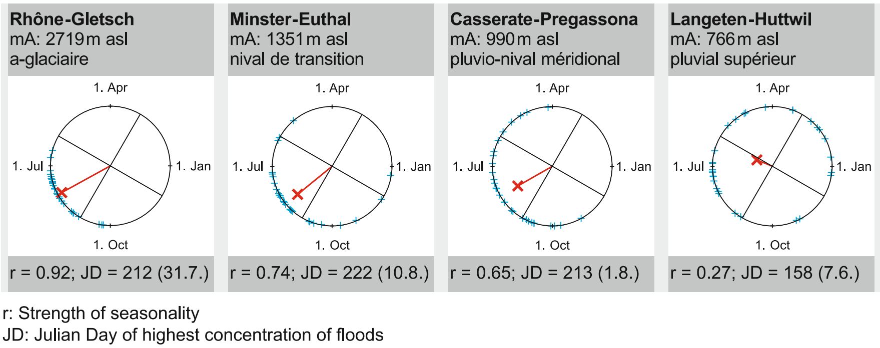


Fig. 18 Seasonal concentration of highest annual flood for various catchments in Switzerland [64]. r = variability of occurrence between 0 (even seasonal distribution of flood) and 1 (flooding always occurs on the same day). r is represented as a vector (*red*). D = mean date of occurrence (*red cross*)

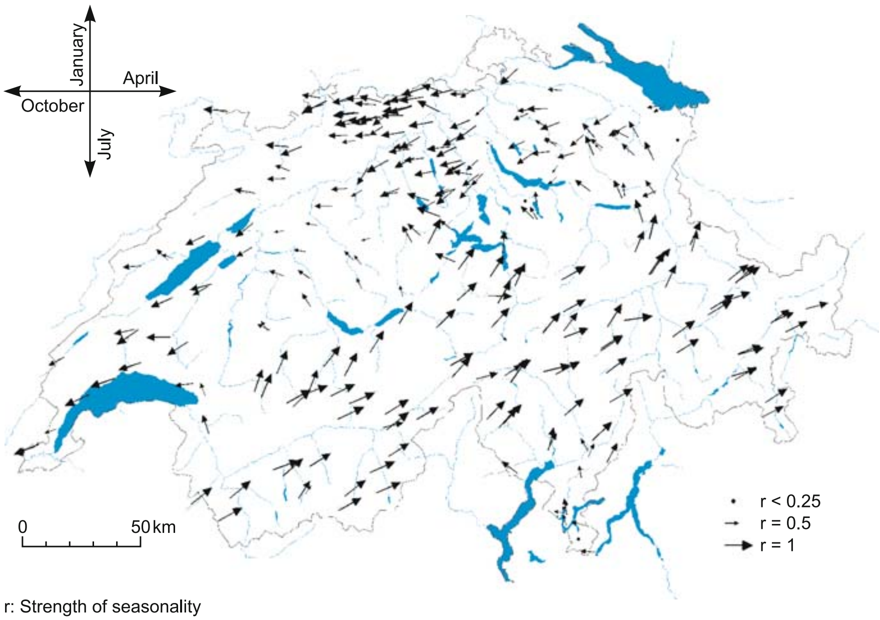


Fig. 19 Spatial representation of low water seasonality for selected runoff measurement stations [85]

on the region (cf. Fig. 19). As a general principle, there is a close correlation between the regime type (Sect. 3.2.2) and the seasonal occurrence of low water periods. In the alpine catchments the lowest flow rates usually occur in the winter half-year, when precipitation is stored in the form of snow.

On the north side of the Alps, the absence of low water during dry periods in summer is primarily attributable to snow and icemelt in the spring [65].

In inner-alpine areas – e.g. the Valais or the Tyrol – drought in summer can also produce low waters. Here drought can affect an entire valley side or even an entire region. The characteristic attributes of alpine catchments, particularly the low storage capacity of soil and subsoil, promote the occurrence of dry periods. This fact has long been recognized, and for centuries most surfaces have had to be irrigated by means of cleverly designed systems [66].

As mentioned several times above, natural low water characteristics in many alpine catchments are influenced by the effects of hydropower usage. Depending on the situation, entire sections of rivers can remain dry for several months, or their runoff regime can be severely altered, with the associated ecological effects (cf. Sect. 3.2.2). In Switzerland the Water Protection Law which came into force in 1991 requires a minimum residual water flow to be maintained in the affected watercourses. The residual flows are determined based on Q95, i.e. flow rates which are attained or exceeded on an average of 347 days (95%) per year. As evidenced by various studies, the legal minimum residual flows represent the absolute minimum from an ecological standpoint [67]. Solutions must be formulated that enable maximum ecological effectiveness and minimal constraints on production. The lessons which were learned in the European alpine region with regard to the hydrological impact of hydropower usage and the maintenance of optimal ecological effectiveness, must be transferred to other alpine regions where the development of hydropower is often the main priority.

4 Water Storage and Water Balance

4.1 Introduction

The water balance describes the elements of the hydrological cycle in a specific area over a specific period. It therefore provides an overview of the water resources which are available over the long term. The classic water balance equation is:

$$P - R - ET - \delta S - I = 0$$

Where

P Precipitation

R Runoff

ET Evapotranspiration

δS Storage change

I Natural subsurface inflow and outflow

The determination and attributes of the water balance elements precipitation and runoff have already been described and discussed in Sects. 2 and 3. The following takes a look at the elements evapotranspiration (ET) and storage change before describing the water balance per se.

4.2 Evapotranspiration

Evapotranspiration is a combination of two processes: evaporation und transpiration: Evaporation covers all water which is directly evaporated from open water surfaces (lakes, rivers, and streams etc.) or wet surfaces (roads, wet vegetation etc.) or sublimated from snow and ice cover. Transpiration covers water which is transported by root plants from the ground to their leaves, and from there is transferred to the atmosphere by the stomata.

Evapotranspiration plays a particularly important role since it links the hydrological cycle and energy budget of the atmosphere. For every kilogram of water that evaporates, 2.446 MJ of energy must be provided at an air temperature of 20° (latent heat of vaporization). This energy is either provided by net radiation or by the environment (atmosphere, soil) in the form of sensible heat. When evaporation condenses on cold surfaces (forming dew), the corresponding latent heat of vaporization is returned to the surrounding area.

Both processes – evaporation and condensation – are of particular importance in the mountains [68]: If snow- or ice-covered surfaces exhibit sublimation conditions due to extremely dry air, the energy used for sublimation is not available for melting. Since only around 12% of the sublimation energy (2.79 MJ kg^{-1}) is required for melting purposes (0.34 MJ kg^{-1}), the same amount of energy can either melt 8 mm of snow cover or sublimate only 1 mm. Conversely, particularly damp and windy conditions produce large volumes of energy available for the snow cover or glacial ice due to condensation: each millimeter of condensed water causes an additional melt of 7 mm of water equivalent snow or ice.

Evaporation is influenced not only by the amount of energy available but also by the amount of water available. If unlimited volumes of water are available in the soil and on surface areas, this is referred to as potential evapotranspiration, i.e. the highest possible evapotranspiration under given climatic conditions. The actual evapotranspiration is the evapotranspiration which can effectively be observed. This is always lower than potential evapotranspiration and is dependent on water availability, plant and surface attributes, net radiation, air humidity, and wind speed. In mountains with a great deal of barren soil and large areas of debris and rock cover which are unable to store water in any great quantities, and with fast-flowing water over steep terrain, actual evapotranspiration is very often limited and hence much lower than potential evapotranspiration.

4.2.1 Calculating Evapotranspiration

This description suggests that it is extremely difficult to measure actual evapotranspiration directly. Indeed, for individual points this can only be done using costly micrometeorological measurements in the lowest layer of the atmosphere, or with the aid of weighing lysimeters that enable evapotranspiration to be determined on a small area of plant-covered ground using the water balance equation: $E = P - R - \delta S - I$. The storage change δS is measured by weighing, the measured gravitational water at the lysimeter outlet corresponds to runoff, and precipitation is separately measured next to the lysimeter. There are no lateral underground inflows and outflows ($I = 0$).

A very similar process is used to calculate evapotranspiration in a hydrological catchment with well-defined hydrogeological conditions. Viewed over longer intervals (one or more years), $\delta S = 0$ can be assumed. $I = 0$ can also be assumed because the area is hydrogeologically verified. Thus, it is easy to calculate evapotranspiration by carefully measuring the areal precipitation (cf. "Precipitation") and runoff. Care must be taken to ensure that all errors in the calculation of individual components of the water balance equation are reflected in the result, for example evapotranspiration.

If is frequently necessary to calculate evapotranspiration on the basis of climate and land use data. On the basis of the calculation of potential evapotranspiration, various procedures have been proposed with the aim of calculating actual evapotranspiration. There are a number of methods [68, 69].

4.2.2 Distribution of Evapotranspiration in the Alps

In the Alps [5] used an extensive range of water balance computations to determine evapotranspiration between 1931 and 1960 for an area covering 195,000 km^2 at an average altitude of 1,270 m ASL. The average value was 540 mm a^{-1}, although an increase from North to South was identified and the individual values vary widely depending on the region, with more than 700 mm a^{-1} recorded in the southern Alpine periphery and only 100 mm a^{-1} in the high alpine region. The reduction in evapotranspiration with increasing altitude fluctuates between 17 and 19 mm per 100 m of altitude.

A high-resolution spatial evaporation map covering the period 1973–1992 exists for Switzerland, which was calculated using modern evaporation models [49]. It shows major variability, on the one hand due to the large differences in altitude, but on the other hand also due to the spatial diversity of the vegetation. This variability is clearly visible in cross-sections through the Alps (Fig. 20). An average evapotranspiration of 484 mm a^{-1} was calculated for Switzerland as a whole (median altitude 1,300 m ASL). The average altitude gradient is around 22 mm per 100 m.

In Austria, which is significantly lower lying (cf. Table 3), the evapotranspiration amounts to 510 mm a^{-1}. On average the actual evapotranspiration is 90% of the potential evapotranspiration for the whole of Austria [70].

Since large areas of the Alps are virtually free of vegetation, evaporation is comparatively low. For the whole of Switzerland, evaporation was calculated at 156 mm a^{-1} for ice and firn, and 234 mm a^{-1} for rock cover [49]. Traffic areas also exhibit low evaporation (199 mm a^{-1}). By contrast, settlements and industrial areas (at 434 mm a^{-1}), agricultural and alpine farming areas (at 436 mm a^{-1}), and in particular forests (at 616 mm a^{-1}) exhibit much higher evapotranspiration, with lakes and rivers registering the highest average of 901 mm a^{-1}. The aforementioned areas also differ substantially over the course of the year (Fig. 21).

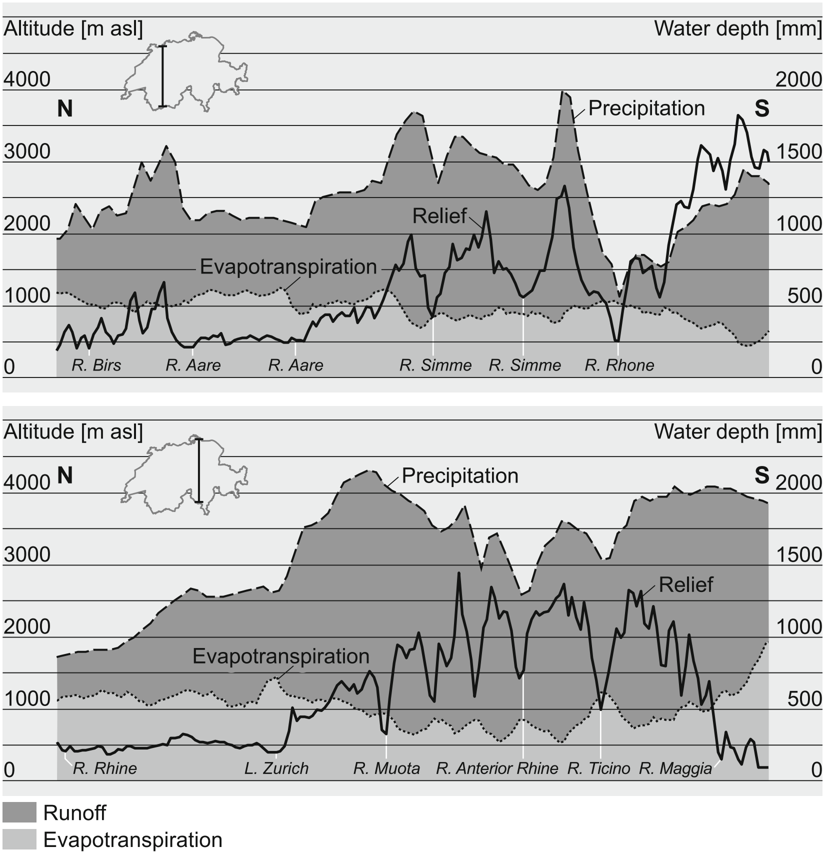


Fig. 20 North–south profiles indicating variation of mean annual precipitation, actual evapotranspiration and runoff (1973–1992) compared to relief. Profile 1 from Basel to Sion (Valais), profile 2 from Schaffhausen to Ascona (Ticino) [27, 29, 49]

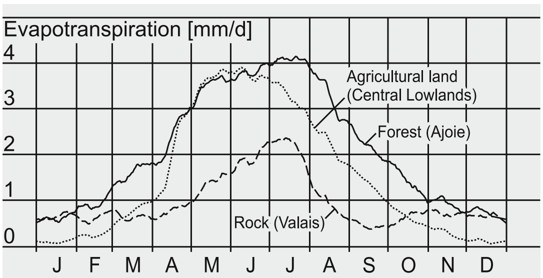


Fig. 21 Examples of mean daily evapotranspiration from varying surfaces [29, 49]

The development of evapotranspiration in Switzerland over time is presented in Fig. 2 of "Impact of Climate Change on Water Resources" [88]. A marked increase in evapotranspiration can be observed since the beginning of the twentieth century.

This is primarily attributable to rising temperatures but may also be due to the increase in agricultural productivity as a result of improved farming methods and fertilization, which has simultaneously increased the volume of transpiration.

4.3 Storage Changes

Water storage and changes in water storage are important elements in the water balance. Water storage provides a balanced supply of water for evapotranspiration and runoff over the short, medium, and long term.

Short-term storage of water for several hours can take the form of interception on the vegetal cover or depression storage in surface puddles. Water is stored over a slightly longer period (days – weeks) in the upper and lower soil. Artificial reservoirs can store water for weeks or even months. However, snow cover or groundwater storage levels as well as lake levels can also be subject to fluctuations that extend over several months (cf. Fig. 22). Glaciers as well as large groundwater reserves located deep underground are capable of storing water for years or even decades. Glacial fluctuations are discussed in Table 1 of "Impact of Climate Change on Water Resources" [88].

The effects of storage are reflected in the runoff regime (cf. Sect. 3.2.2). These storage depots help to offset shortages during dry periods. On the other hand, snow

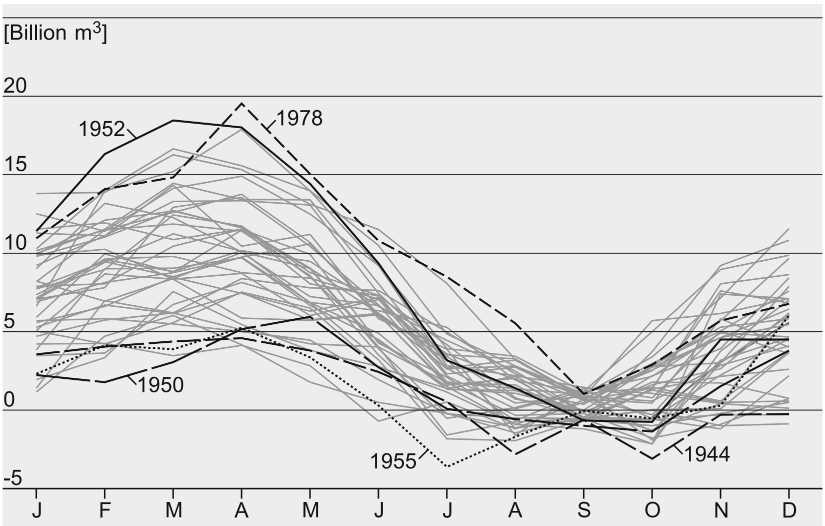


Fig. 22 Monthly variations in total water storage volumes in Switzerland from 1940 to 1981. Water storage volumes include medium-term changes in soil moisture, groundwater, snowpack, lakes, and reservoirs. Long-term changes in groundwater and glaciers are not taken into account [71]. Minimum and maximum years are indicated

stored in large quantities can also pose a threat if it melts rapidly at times of heavy precipitation.

Water storage in soil, groundwater, and snow cover is extremely difficult to quantify, whereas changes in lake and reservoir levels can be measured and calculated with relative ease. Figure 22 charts the total short- and medium-term storage volumes in Switzerland based on comprehensive water balance calculations [71]. Strong seasonal fluctuations in storage volumes are immediately identifiable. These are attributable not only to natural snow cover during the winter, but also to man-made reservoirs, which in Switzerland have a maximum storage capability of almost 4 billion cubic meters of water. Natural lakes play an important role in mitigating peak discharge. During flood periods, short-term fluctuations in the water level of lakes may be temporarily relevant. Today these fluctuations are generally much less severe than at the end of the nineteenth century [71], since most lakes are now artificially managed in order to safeguard against excessively low or high water levels.

When compared against the figures in Table 4, the maximum storage levels shown in Fig. 22 (with a peak in spring of almost 20 km^3) are striking, particularly given the fact that Fig. 22 shows the monthly averages rather than the maximum and minimum levels. Moreover, it is also worth remembering that this volume of stored water is replenished year after year. These impressive statistics also clearly explain why the Alps – like many other mountainous regions in the world – are of such importance for neighboring lowlands (cf. Sect. 5).

An analysis of the figures in Table 4 gives rise to some interesting comparisons. For example, the storage volume in reservoirs is the same as the average volume of water stored by snow cover on 1 April. While the water volume stored by snow cover is much larger than that contributed by glaciers, even in hot years, it is not available in the same months. Drinking water consumption accounts for only 2% of water supplies in Switzerland. Only a very small proportion of water is used for irrigation. However, these statements are generalizations, and conditions can vary widely depending on the region and season, thereby changing the impact of the individual components.

Table 3 Water balance in the Alps [5], Austria [87], Switzerland, and some of its alpine fluvial basins [51]

Region	Period	mA	Area	P	R	ET	δS
		[m ASL]	[km^2]	[mm]	[mm]	[mm]	[mm]
Alps	1931–1960	1,270	195,500	1,450	910	540	
Austria	1961–1990	770	83,875	1,144	634	510	
Switzerland	1961–1990	1,312	41,285	1,458	991	469	−2
Rhine-Chur	1961–1990	1995	3,270	1,465	1,123	343	−1
Rhone-Léman	1961–1990	2084	5,458	1,435	1,034	407	−6
Ticino	1961–1990	1483	3,352	1,943	1,458	485	0
Inn	1961–1990	2334	1,818	1,248	964	286	−2

4.4 Water Balance

To calculate the water balance it is necessary to obtain an overview of the various components in order to estimate their relevance and variability over time and space. Since it is not easy to determine the individual components, the water balance equation

$$P - R - ET - \delta S - I = 0$$

will never work out exactly. Often, therefore, one of the parameters is calculated as a remainder to which all errors are attributed. Alternatively, an attempt can be made to balance out the errors, as in the example of [5, 31]. Comparisons of different water balances are also difficult due to the fact that the climate is constantly changing. Various methods and intervals are applied.

The overview of water balances in the Alps shown in Table 4 prompts some interesting comparisons. While the overall figure for evaporation in the Alps is 37% of precipitation, the percentage of evaporation in Austria is 45% compared to only 32% in Switzerland. This is because it rains far less in Austria due to the country's more continental climate, and the country's lower altitude gives rise to more evaporation. Even within Switzerland there are major regional differences. Despite being situated at a much higher altitude, the Rhone, Rhine, and Inn basins record no higher precipitation than the average for Switzerland due to the inner-alpine location of these regions. Yet the volume of precipitation in the Ticino basin, in the south of Switzerland, is substantial.

These regional effects in the Rhone area (Valais with low precipitation and high evaporation) or in the Ticino area (high precipitation and low evaporation) are clearly expressed in the transversal profiles shown in Fig. 20. This diagram also

Table 4 Water storage volumes in Switzerland compared to flows in the water balance and water applications (from [86])

Storage volumes in Switzerland		
Lakes	130	km^3
Glaciers 1850	100	km^3
Glaciers 2007	45	km^3
Reservoirs	4	km^3
Useable Groundwater	11	km^3
Snow cover on 1 April (1961–1985)	5	km^3
Water flows		
Precipitation (1961–1990)	60	$km^3\ year^{-1}$
Evapotranspiration (1961–1990)	19	$km^3\ year^{-1}$
Runoff from Swiss area (1961–1990)	40	$km^3\ year^{-1}$
Total runoff from Switzerland (1961–1990)	53	$km^3\ year^{-1}$
Storage in glaciers 1974–1981	0.4	$km^3\ year^{-1}$
Melting of glaciers 1998–2006	0.9	$km^3\ year^{-1}$
Drinking water consumption 2005	1	$km^3\ year^{-1}$
Irrigation during drought summer	0.1	$km^3\ year^{-1}$
Water for artificial snowmaking	0.008	$km^3\ year^{-1}$

illustrates the major surplus of precipitation over evapotranspiration (green areas) which gives rise to the enormous resources of water in the Alps (cf. also Sect. 5).

The water balance is not only subject to significant regional and seasonal fluctuations but also to longer-term changes associated with climate change. Fig. 2 in "Impact of Climate Change on Water Resources" [88] charts the changes in the elements of the water balance in Switzerland since 1901. It shows the major fluctuations to which precipitation is subject, and suggests that precipitation has increased slightly over the years. At the same time, however, evapotranspiration has also been steadily increasing in line with rising temperatures. All in all, therefore, runoff – and hence water resources in Switzerland – appears to have remained virtually unchanged over the past 100 years. It would be a mistake, however, to assume that this trend will continue, since regional and local changes in the water balance have been identified and further changes must be expected (cf. "Impact of Climate Change on Water Resources" [88]).

5 The Alps: Europe's Water Tower

5.1 Introduction

Mountains and highlands typically produce substantial volumes of water. The reason for this is that the air masses are forced to rise and subsequently cool, releasing humidity in the form of precipitation (orographic precipitation, cf. Sect. 2.2). Furthermore, evapotranspiration is reduced due to a lower rate of net radiation, lower temperatures, more frequent snow cover, and a shorter vegetation period. The discharge formed in mountain regions is subsequently transported to adjacent lower-lying areas via river systems. Downstream, the mountain water performs an important function for irrigation and food production.

One key contributor to the importance of mountains in their role as water towers is their ability to store winter precipitation – temporarily or over a longer period – in the form of snow and ice, which melt only in spring and summer, i.e. precisely when the water supply in the lowlands is at a minimum and agricultural demand for water is high. Moreover, the essential mountain contributions in summer are highly dependable. It is therefore generally agreed that mountain regions, with their disproportionately high discharge compared to lowlands, are of significant hydrological importance.

5.2 The Significance of the Alps for Downstream Hydrology and Water Resources

The Alps exhibit the general hydrological features typical of mountain areas. Additionally, they are exposed to the influence of seas in three directions (the

Table 5 Contribution of the alpine area to total discharge, respective shares in area and corresponding disproportionalities (annual mean and maximum monthly average)

	Rhine	Rhone	Po	Danube
Mean annual contribution to total discharge (%)	34	41	53	26
Maximum monthly contribution to total discharge (with month of occurrence) (%)	52(VI)	69(VII)	80(VIII)	36(VIII)
Share in total area (%)	15	23	35	10
Disproportional influence of the Alps (annual)	2.3	1.8	1.5	2.6
Disproportional influence of the Alps (maximum monthly)	3.5	3.0	2.3	3.6

Atlantic to the west, the North Sea to the north and the Mediterranean to the south) and lie in a zone of predominantly westerly winds. This enables sizeable amounts of humidity to be transported and subsequently extracted from the atmosphere. Because of their significantly higher annual discharge compared to the surrounding lowlands, the Alps are often referred to as the Water Tower of Europe [72]. With a mean contribution of 34% of the total discharge, the alpine regions of the River Rhine supply 2.3 times more water than might be expected on the basis of surface area alone. Table 5 lists the corresponding figures for all four major alpine rivers of Switzerland. The maximum contribution of the Alps occurs in the summer months, ranging from 36% (Danube, in August) to 80% (Po, in July). Since this summer runoff from the Alps originates from snow- and icemelt, it is highly reliable and mitigates the variability of the precipitation-driven flow characteristics downstream.

Figure 23 illustrates the high hydrological productivity of the Alps using specific discharge figures, i.e. discharge per unit area. For the Rhine and the Rhone Rivers, a clear pattern is evident with high specific discharges in the upper sections (typically 40–70 $\mathrm{l\,s^{-1}\,km^{-2}}$ or higher) and a steady decline as the catchment area grows in size and the influence of the Alps declines. The influence of the Alps is particularly visible in the Danube catchment which originates outside of the Alps and is subject to alpine influence via the River Inn (Fig. 23, Danube at a catchment size of about 75,000 $\mathrm{km^2}$).

5.3 The Global View

Building on the above concepts developed for the European Alps, studies were also conducted for other mountain regions worldwide.

A first approach was based on discharge gauge data from 20 case studies around the world to produce a representative assessment of mountain waters in different climatic conditions [73]. A study of the hydrological characteristics of these basins

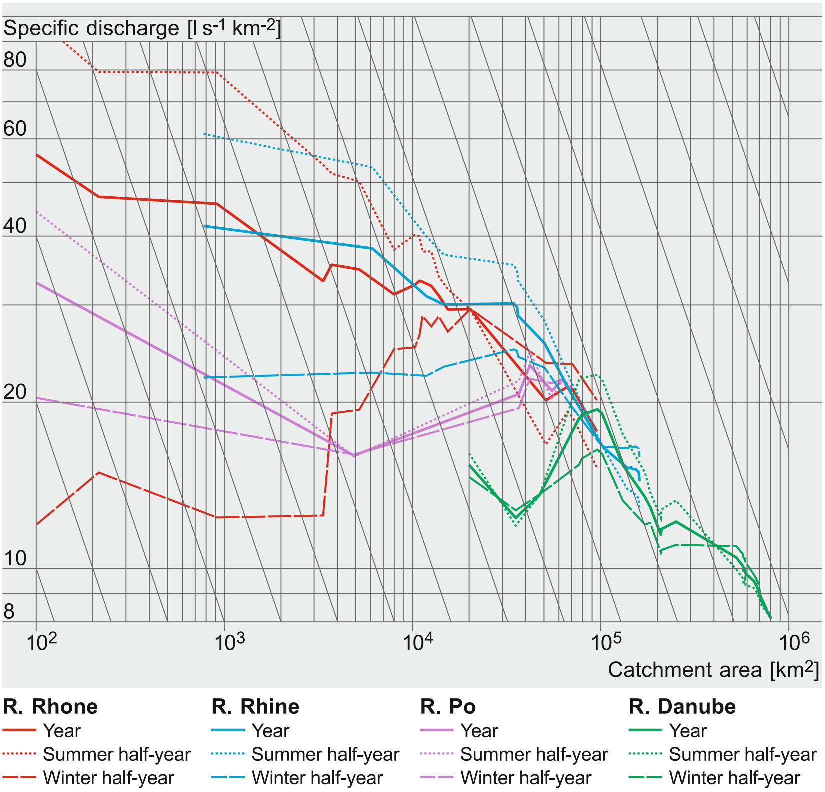


Fig. 23 Specific discharge for the four main alpine rivers Danube, Rhine, Rhone, and Po, annual mean as well as summer and winter half-year mean values

showed that the mountainous sections of rivers usually produce disproportionately high amounts of discharge. While mountain regions in humid climate conditions – such as in the Alps – produce about twice the discharge expected on basis of their share of the total basin area, mountain runoff may constitute 50–90% or more of total discharge in semi-arid and arid areas: the drier the lowlands, the greater the importance of the relatively humid mountain areas [74]. Since the runoff originating in mountains has a low year-to-year variability, its occurrence is highly reliable. Furthermore, it usually coincides with the dry period in the lowlands during summer.

The first spatially distributed assessment of the hydrological significance of mountains on a global scale was recently conducted by [75] on the basis of a hydrological model and a digital elevation model. Among other things, this analysis showed the disproportionately high runoff of mountain areas for the entire global land surface at a resolution of 0.5° × 0.5 (Fig. 24). Furthermore, by incorporating climate and population data, it found that more than 50% of mountain areas play an essential or at least supporting role for the water resources in the respective downstream regions.

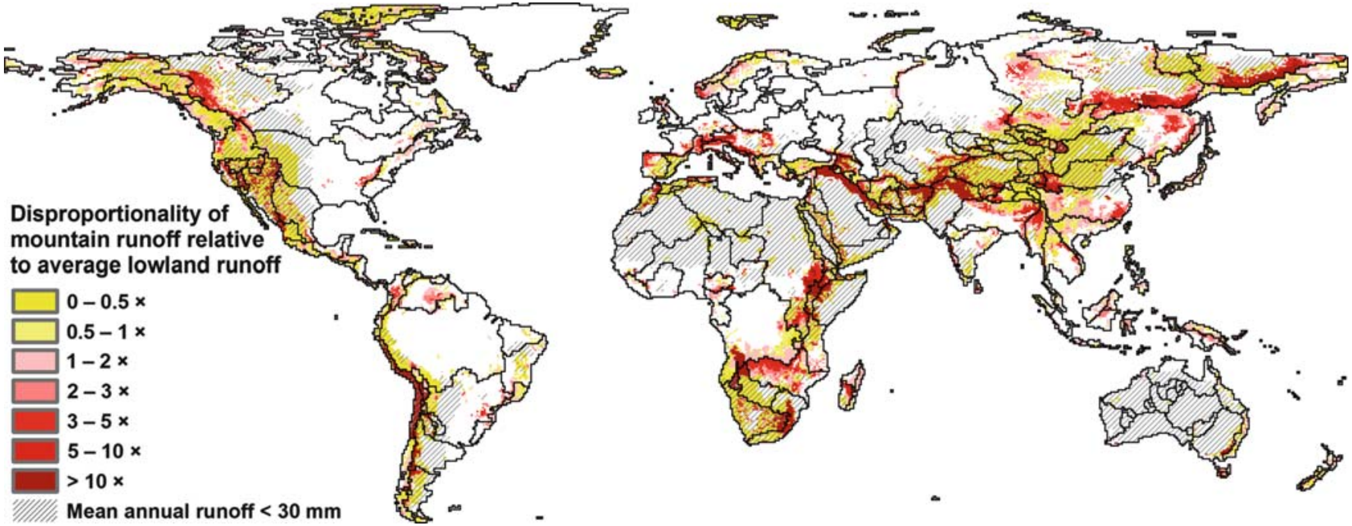


Fig. 24 Disproportionality of mountain runoff relative to average lowland runoff for mountain areas at $0.5 \times 0.5°$ resolution

5.4 The Future Role of Mountain Water Resources

5.4.1 Global

Since mountain regions are highly vulnerable to climate change [76], marked changes in snow and ice occurrence are to be expected. Since this means an alteration in seasonal storage rates, these changes will also affect seasonal discharge patterns.

Besides global climate change, mountain water resources are also influenced by demographic factors: population growth in critical lowland areas aggravates potential water scarcities and increases the pressure on streamflow originating in mountain areas. The growing demand for food and changes in dietary habits (e.g. increase in meat consumption) have the same effect. Furthermore, the future will see more intense competition for water for use in hydropower generation and industry.

The role played by changes in land use is less certain to predict. While population growth in mountains is likely to lead to intensified land use, with the resultant local soil degradation and erosion, the effects on downstream water resources are likely to be less marked.

5.4.2 Alps

As far as the European Alps are concerned, at first sight the situation does not appear too critical due to the abundance of water. The highest average areal precipitation figures, as observed in the Central Alps (Gotthard region), exceed 2,300 mm a^{-1} and the corresponding runoff is around 2,000 mm a^{-1}, equivalent to a mean specific runoff of approximately 70 l s^{-1} km^{-2} [77].

However, changes in the seasonality of snowmelt coupled with shifts in precipitation patterns may result in water shortages in the summer and autumn, especially in downstream regions. For the River Rhine, for instance, some scenarios show a decrease in total discharge of more than 50% at the German-Dutch border for the low-flow period in the autumn [78]. Moreover, the declining impact of snow and ice melt will increase the year-to-year variability of discharge in the summer and autumn.

5.5 Need for Knowledge

As the world's water towers, mountains will continue to play an essential role in meeting increased demands for food, drinking water, energy supplies, and industrial production in the twenty-first century. Thanks to their specific climatic and hydrological characteristics, mountains play a key role in the global water cycle. Water management must therefore start in mountain regions. Given the above considerations, it is essential to emphasize the importance of increasing our knowledge about mountain water resources [79]. Over the past decade the growing volume of available data, combined with the use of hydrological and climatological models, has driven progress in this field. However, due to the remoteness of the study area and the major strain placed on gauging devices, mountain hydrology is still a challenging field. While this is especially true for mountains in developing and emerging countries, reliable and long-term measurements at high altitude areas are relatively scarce even in the densely gauged Alps.

Further increasing our knowledge about mountain water resources will help us to understand the interaction between mountains and lowlands. It is this interaction that needs to be accorded high priority in terms of watershed management and in the interests of mitigating any conflicts that may arise over the distribution of this precious resource.

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Impact of Climate Change on Water Resources in the Alpine Regions of Switzerland

Bruno Schädler and Rolf Weingartner

Abstract Alpine regions have been and will be responding sensitively to global climate change compared to other European regions. This chapter analyses past and future changes in the climate of the Alps and its consequences on the elements of the water cycle. One obvious consequence is the melting of European glaciers by more than 65% since the end of the Little Ice Age in 1850. As a result of temperature increase hydrological regimes are changing. How will climate and water balance change in the future? Scenario results for the European Alps and in more detail for the Swiss Alps are discussed. Consequences in inner Alpine valleys are less precipitation in summer and hence more droughts.

Keywords Climate change, Glacier, Long time series, Precipitation, Runoff

Contents

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1 Introduction

The importance of mountains in their role as headwaters and sources for the water resources of lowlands has long been recognized – particularly by the scientific community (cf. [1]). But it has also increasingly become the focus of political attention since the debates at the 1992 Rio Earth Summit and even more so during the International Year of Mountains in 2002. With regard to hydrology, the symbolic term “water tower” was coined [2] and has now been widely adopted. It expresses the important role that mountains play in providing fresh water for adjacent areas downstream [3]. Changes in water resources in mountain areas will obviously have implications for water resources in the lowlands. Specific climate changes in headwaters will affect the availability of water for the lowland population which might experience different kinds of climate changes.

Man-induced climate change is not just a scenario for the future; it has been experienced since the beginning of industrialization, which coincided with the so-called Little Ice Age in the mid-nineteenth century. This contribution analyses the changes in climate and especially in the hydrological cycle and water resources mainly over the last 150 years and for the Swiss Alps. Fortunately, in Switzerland and therefore in the Swiss alpine region, climate and water resources have been under observation for this entire period.

Thanks to the establishment in 1823 of the Swiss Association of Natural Sciences (now the Swiss Academy of Sciences), the first systematic monitoring of the climate was started at that time, with the Glaciological Commission in charge of Swiss glaciers, the Meteorological Commission in charge of climate observation, and the Hydrological Commission responsible for observing lakes and rivers. Within a few years national state agencies were established, charged with ensuring the long-term systematic observation of the climate and the hydrological cycle. These include the Swiss Meteorological Institute and the Swiss National Hydrological Survey, which is now a department of the Federal Office for the Environment. Glaciers continue to be observed by the Swiss Academy of Sciences.

2 Temperature and Precipitation

Temperature and precipitation are the climate factors that exert the strongest influence on the hydrological cycle. Both have been under observation for a very long time. Climate Change in the Alps over the last 250 years has been extensively studied [4]. Monthly homogenized records for time series dating back to 1760 for temperature and back to 1800 for precipitation have been compiled. After a decrease of 1°C between 1790 and 1890 an increase of about 2°C until the early twenty-first century has been observed. The warming process over the twentieth century was more accentuated in summer than in winter. During the last 25 years the winters and summers have warmed at comparable rates, which is not typical for the regional climate evolution over the past 250 years. In [5] two antagonistic centennial trends are documented with a wetting trend (since 1860) in the northwest of the Alps and a drying trend (since 1800) in the southeast.

In Switzerland there are 12 stations that cover the period from 1864. Seven stations are located in the lower areas below 1,000 m ASL, four between 1,000 and 2,000 m, and the highest is at 2,490 m ASL. Since the exact location was moved and instruments were changed on several occasions, it was necessary to homogenize all but one time series [6]. Most of the yearly and seasonal temperature series show a significant positive trend. The gradients of the yearly series range from 0.9°C/100 years to 1.2°C/100 years on the northern side of the Alps and the central valley, while on the southern side of the alpine main ridge smaller gradients of 0.6°C/100 years are observed. In terms of seasonal values, the largest increases are found in autumn for stations at higher elevations (0.8–1.3°C/100 years) and in winter for stations at elevations of $<$1,000 m ASL ranging from 0.9°C/100 years to 1.6°C/100 years [6]. Compared to the temperature trend of the global mean at 0.42°C/100 years and the land surface of the northern hemisphere at 0.63°C/100 years [7], the overall trend in the Swiss Alps is about 1.4–2.4 or 1.0–1.6 times higher. Clearly, the times series do not show a continuous increase in temperature but rather two distinct jumps at the beginning and close to the end of the twentieth century. A linear trend of 5.71°C/100 years as a mean value in Switzerland for the period 1975–2004 has been identified [8], which is more than twice as high as for the northern hemisphere.

In terms of precipitation, significant trends are observed for yearly values only at four stations in the lower areas: the slopes of the trends range from +7 to +10% per 100 years [6]. As for the seasonal values, significant precipitation trends are found at all but the southern stations only for the winter season. The strongest trends are observed in the western part of Switzerland, at +35 to +37% per 100 years. For the other stations, the values amount to +16 to +25% per 100 years. However, based on a progressive analysis of the yearly and winter series, it becomes clear that the trends are strongly influenced by the period of time, with higher precipitation volumes starting between 1940 and 1950, becoming even more pronounced around 1970 [9]. These findings are comparable with the results of other studies despite the fact that these studies analyzed different time periods and different data sets [10–13].

Risk trends in heavy precipitation are important for assessing changes in natural hazards. In a comprehensive study [13], the 100-year (1901–2000) time series of 104 rain gauges in Switzerland was analyzed. A clear trend signal was found for winter and autumn for all statistics related to precipitation strength and occurrence. The centennial increase is between 10% and 30% for the high quantiles and the seasonal 1- to 10-day precipitation. The winter trend signal is strongest in northern and western Switzerland (Fig. 1). However, these trends are no longer valid if we look further back into the past. Hegg and Vogt (2005) [14] analyzed yearly maximum daily precipitations measured at 18 stations in Switzerland from 1864 to 2002 and found that, in the period between 1890 and 1970, the maximum yearly 24 h-precipitation was mostly lower than before or after this period. Consequently they found a positive trend for the entire period for only one station (in the southern part).

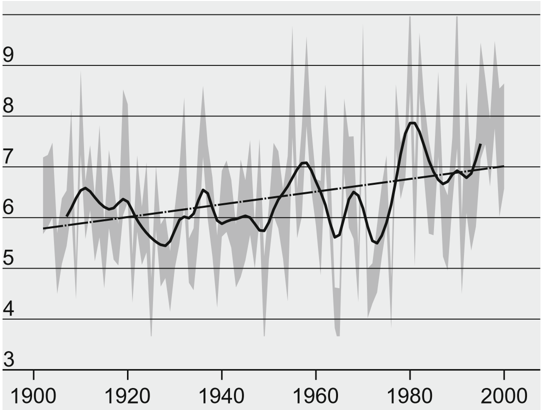

Fig. 1 Time series of precipitation intensity (mean wet-day precipitation in mm/d) from 38 stations in northern Switzerland during the winter. The *blue curve* denotes the lower and upper quantile of the station values. The *bold line* depicts the low-pass filtered (11-point binomial filter) median of all station values. Trends (denoted by the *straight red line*) are estimated from the time series of medians in the station pool. Figure from [13]

3 Snow and Ice

Seasonal snow cover strongly impacts runoff regimes in mountainous regions. During the winter season the snow pack grows in many prealpine and most alpine regions, whereas it melts in the spring and summer season up to altitudes of about 3,000–3,500 m ASL, depending on exposure. Time series of 190 observing stations between 275 and 2,540 m ASL have been analyzed for the period 1931–1999 [15]. Even if a considerable interannual and spatial variability of snow parameters is observed, the mean snow depth, duration of continuous snow cover and the number of snowfall days show very similar trends. No uniform trend is discernible over the whole period: a gradual increase until the early 1980s is followed by a statistically significant decrease towards the end of the twentieth century. The latter finding is confirmed by an analysis of 27 long-term water equivalent time series, which showed that annual maximum water equivalent values have been decreasing for most stations since the 1980s. Generally speaking, relative trends are negligible for altitudes above 1,300 m ASL, whereas major negative trends of up to –2.5%/year are found for lower stations (500 m ASL). Model analysis has shown that, in general, the shift in temperature explains most of these trends [16].

Water is stored over relatively long time periods in alpine glaciers. During the so-called Little Ice Age, the last maximum glacier stage was reached around 1850. In Switzerland about 110 billion cubic meters of ice (see Table 1) was stored, at that time covering about 1,800 km^2 (4.3%) of Switzerland's territory. Between 1850 and 2000 about 40% of the glacier surface and around 50% of the ice volume disappeared due to the rise in temperature [17]. These figures are comparable to the changes in ice in the entire European Alps, where initially 200 billion cubic meters of ice were stored under a surface of around 4,475 km^2. By 2000, shrinkage in the European Alps was more pronounced due to the lower mean altitude compared to the Swiss Alps, with surface area diminishing by 50% while volume losses

Table 1 Water storage in European alpine glaciers from the Little Ice Age to the present [17–20]

Year	European Alps		Switzerland	
	Ice (10^9 m^3)	Water (10^9 m^3)	Ice (10^9 m^3)	Water (10^9 m^3)
1850	200	182	110	100
1973	100	91	75	68
2000	75	68	55	50
2006	67	61	49	45

amounted to 62% [18]. Because of the exceptionally warm summer experienced in Central Europe in 2004, another 10% or so of the ice volume has disappeared [19, 20].

4 Water Balance and Runoff

Changing climate, with rising temperatures and changes in precipitation regimes, has had a distinct influence on runoff regimes and the water balance. Starting in 1931, Birsan et al. (2005) [21] analyzed daily stream flow records of 13 near-natural watersheds in Switzerland. They identified a change in runoff regimes toward higher runoff in winter and lower runoff in the other seasons. For the shorter period 1971–2000 the trend in 49 time series was positive for yearly values and all seasons, although still most pronounced in winter, due mostly to an increase in the high runoff quantiles or even at times in the maximum runoff. This last finding is comparable to the above-mentioned increase in precipitation in many parts of Switzerland during the winter season. A statistical analysis clearly indicates a correlation of positive runoff trends with mean basin elevation and glacier and rock coverage. This relationship suggests that – besides precipitation changes – changes in snow and ice distribution resulting from temperature change exerts the strongest influence on the shift in the regime of individual watersheds. Obviously, regime types [22] change from ice- and snow-influenced regimes to more snowmelt and rainfall influenced regime types.

Yearly water balance time series since 1901 based on measured data have been established and analyzed by [23–25]. Recently, special importance has been attached to the fact that quantification of areal precipitation in mountainous environment remains a very difficult task. With the aim of achieving high spatial resolution, a specific methodology has been developed based on a comprehensive view of the water balance in different spatial and time scales [26].

For Switzerland the water balance has been established for the four most important river basins (Table 2): Rhine (27,970 km^2; northern part), Rhone (5,220 km^2; western inner Alpine part), Ticino (1,515 km^2; southern part), and Inn (1,944 km^2; eastern inner alpine region of Switzerland) and also for the entire territory of Switzerland (41,287 km^2).

Table 2 Water balance elements for the time period 1901–2000 for four river basins in Switzerland and for Swiss territory

	Precipitation P (mm/a)					Natural runoff R (mm/a)					Evapotranspiration E (mm/a)				
	Switz.	Rhine	Rhone	Ticino	Inn	Switz.	Rhine	Rhone	Ticino	Inn	Switz.	Rhine	Rhone	Ticino	Inn
Mean	1,429	1,404	1,432	1,829	1,215	992	949	1,090	1,421	928	448	461	386	411	302
Standard deviation	201	205	234	342	180	153	165	135	309	130	71	78	90	128	92
Maximum	1,844	1,832	1,931	2,942	1,650	1,312	1,286	1,390	2,336	1,284	625	658	708	754	549
Minimum	859	849	775	1,048	754	629	527	759	779	611	247	251	121	125	96
Trend	1.2	1.18	2.03	−0.22	−0.11	0.07	0.07	0.83	−2.18	−0.42	1.05	1.07	0.87	1.92	0.15
	Storage change (Glaciers) (mm/a)														
	Switz.	Rhine	Rhone	Ticino	Inn										
Mean	−11	−6	−44	−3	−15										

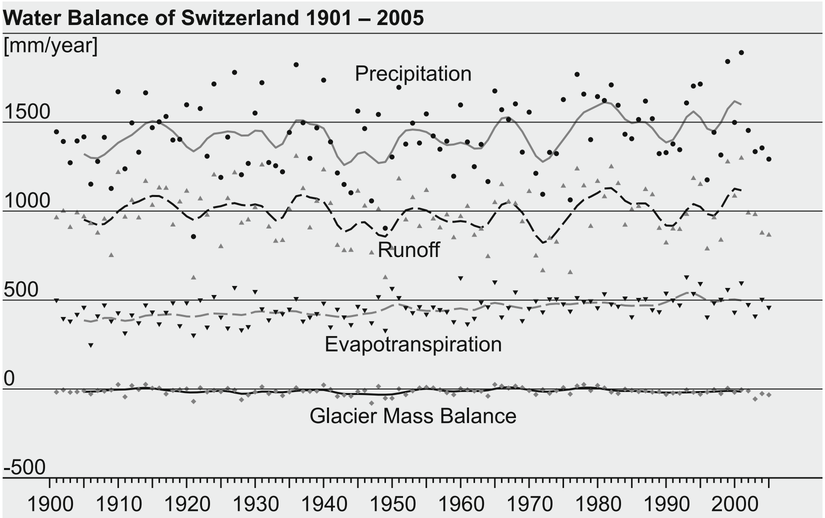

Fig. 2 Water balance of Switzerland. Points are values for the hydrological years; the lines are 9-year low-pass Gaussian filtered values

The amount of precipitation varies widely depending on climate region and topography, with the highest values and highest variability observed south of the Alps in the Ticino basin and the lowest values measured in the inner alpine valley of the Inn, which is protected from winds in all directions. Since precipitation trends are inhomogeneous for seasons and for regions, no significant trends for these basins have been identified over the entire time period, even if they appear obvious from Fig. 2.

In almost all regions, evapotranspiration essentially depends on temperature. Because of abundant precipitation, most of the time it corresponds to almost the same values as potential evapotranspiration. Indeed, a significant positive trend in all but the Inn basin can be observed.

As a result, the distribution of runoff values is similar to the finding for precipitation. For the Rhine basin, runoff values remain constant as the amount of rising evapotranspiration is compensated by almost the same amount of rising precipitation. For the Rhone, the increase in precipitation seems to be higher, resulting – together with the contribution made by melting glaciers – in higher runoff, whereas for the Ticino and also to a less pronounced extent for the Inn, runoff diminishes due to lower precipitation.

The contribution of ice melt to the mean runoff is often overestimated. Even if 53 billion m^3 of glacier ice disappeared in Switzerland during the twentieth century, the contribution to mean runoff only amounts to about 1%. However, for individual small watersheds close to the glaciers, the contribution made by melt water from glaciers is important. Changes here result in the changes in runoff regimes that have already been discussed.

5 Climate Scenarios for the Twenty-First Century in the Alps and their Implications for Water Resources

5.1 *Scenarios for the Alps*

Following the IPCC Scenarios A1B, [27] calculated climate change scenarios using CLM (Climate version of Local Model) runs for the twenty-first century. On the basis of these results [28] summarizes Climate Change for the Alps as follows: A temperature rise of 3.9°C up until the end of the twenty-first century is projected for the Alps, compared with a warming of 3.3°C for Europe as a whole. The warming will be particularly elevated in the high mountains (>1,500 m) with a 4.3°C increase. Until the mid twenty-first century the temperature increase will be comparatively low (1.4°C). Temperature increase would be significantly lower under the lower greenhouse gas concentration scenario B1, with a projected +2.6°C up until the end of the century.

Depending on the region, the decrease in yearly precipitation ranges between −1% and −11% with the strongest decrease in the south-western Alps. When regarding the seasons, the summer shows the strongest changes with strongest effects in the south-west with −41% and the least effects in the north-east with −25%. In spring most regions will receive more precipitation (−3% in SW to +23% in NE). As a result of the changing precipitation patterns there will be a change in the incidence of dry periods (>5 consecutive days without precipitation), particularly for summer (+36%) with the strongest relative increase in the northern Alps (NW +73%), where the number of dry periods is low under present conditions.

A consequence of these changes is an increase of snow line with altitude and a decrease of days with snow cover. Even at altitudes around 1,500 m the models calculate a reduction of the amount of snowfall of about 20% up to the end of the twenty-first century. REMO [29] suggests that below 500 m snow could nearly disappear.

While changes in average yearly runoff totals are in line with precipitation changes, they differ significantly from precipitation changes on a seasonal scale, particularly in winter and spring. In these seasons, the amount of snowmelt and changes in the proportion of precipitation falling as snow are stronger drivers for runoff than changes in precipitation alone. Higher temperatures will not only lead to increased rainfall and less snowfall but also to a higher proportion of snow that will already melt in winter (and not in spring). This could lead to an increase in winter runoff in the Alps by up to 19% as well as a decline in snowmelt in spring by up to 40% with a reduction in spring runoff by up to −17% [29]. In particular, the strong decrease in summer runoff – which for some parts of the southern and central Alps is predicted as being up to −55% by the end of the twenty-first century – could cause problems with water availability within the Alps, and even more in downstream regions (cf. [1]).

5.2 Scenarios for Switzerland

For Switzerland [30] published detailed climate scenarios developed by [31]: Until 2050, the warming will be practically the same on the northern and southern sides of the Alps. According to a middle estimate, the temperature will increase in northern Switzerland by 1.8°C in winter and 2.7°C in summer, and in southern Switzerland by 1.8°C in winter and 2.8°C in summer. For the transitional seasons, the warming will be comparable to the warming in winter (spring: 1.8°C on the northern and southern sides of the Alps; autumn: 2.1°C on the northern side, 2.2°C on the southern side).

Temperature extremes show the most distinct trend. Climate models show a more significant increase in absolute maximum temperatures than in the mean daily maximum. According to this scenario, conditions like those of the 2003 summer heat wave will occur every few decades in the case of medium warming, and every few years in the case of strong warming.

In contrast, the frequency of cold spells and the number of frost days will decline. In winter, the daily temperature variability will become smaller because minimum temperatures will rise more strongly than mean temperatures. This effect will be particularly pronounced in areas where the snow cover decreases as a result of the warming.

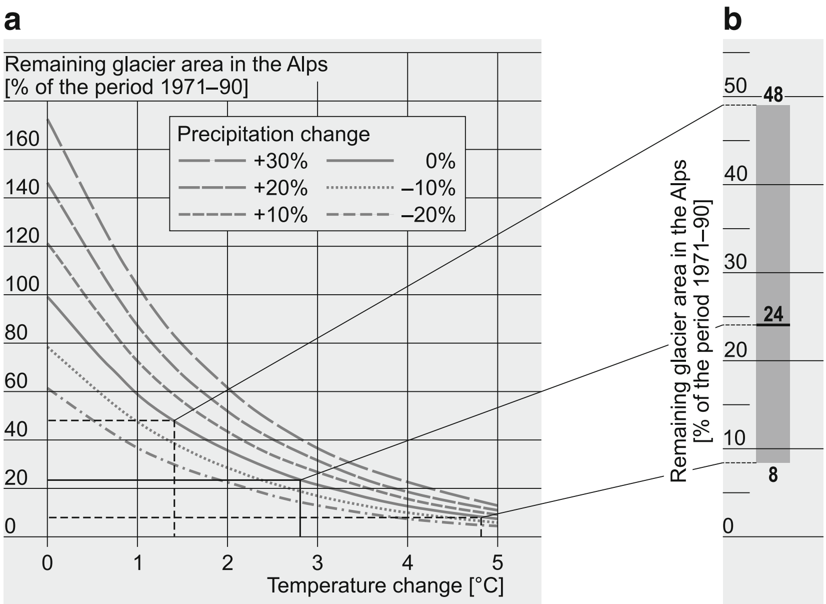

Fig. 3 (a) Change in alpine glaciation with an increase in summer temperature by 1–5°C and a change in annual rainfall between –20% and +30%. (b) According to this scenario, glaciation will decrease by about three-quarters by 2050 [18]

For precipitation the changes in the various regions differ in all seasons by only a few percent. On the northern side of the Alps, an increase of 8% is expected in winter (11% on the southern side) and a decrease of 17% in summer (19% on the southern side) by the middle of the twenty-first century. In spring and autumn, precipitation increases or decreases are possible. In summer, the area of uncertainty is particularly large. As for precipitation extremes model results indicate that heavy precipitation events of a kind that occur only every 8–20 years nowadays will on average occur every 5 years by the end of the century. The situation is less clear for the summer season. Although the models show a distinct decrease in the mean rainfall, the 5-yearly extreme value shows a slight increase.

The retreat of glaciers will be the most obvious change in the Alps as a result of climate change. Model calculations of the expected glacier retreat in relation to the reference period 1971–1990 are shown in Fig. 3a. They were calculated for a warming in summer between +1 and +5°C and a change in annual rainfall between –20% and +30%.

According to the climate scenario here, by 2050, the area covered by alpine glaciers will have diminished by about three quarters in the case of medium warming (Fig. 3b). In the case of a moderate warming, the loss in glacier area will be about 50% and in the case of strong warming about 90%. The relative losses will be smaller than the calculated average change for large glaciers and larger than the average for small glaciers. Many small glaciers may disappear.

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Human Interventions

Bernhard Wehren, Bruno Schädler, and Rolf Weingartner

Abstract In mountainous regions water often constitutes the dominant – and even in many cases the only – useful natural resource. Therefore, water is used for various purposes, especially for the production of hydropower, for water supplies, irrigation, and artificial snow making. The use of water resources and other human interventions, for example structural control and the correction of water courses, may result in fundamental quantitative changes in the natural water cycle or in the natural runoff characteristics on varying time scales. These interventions also significantly influence a wide range of other physical, chemical, and morphological parameters of rivers and streams.

In the future changes in runoff patterns forced by climate change as well as a growing demand for water for various applications in mountainous regions must be assumed. As a consequence, the strain on natural water supplies in alpine regions will be much stronger, especially at the local level. As mountains play an important role as water towers for the lowlands, this may also impact the water supply in the surrounding regions.

Keywords Artificial snowmaking, Hydropower, Irrigation, Runoff characteristics, Water use

Contents

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1 Introduction

During the course of the twentieth century, human use of water grew exponentially – particularly in the alpine region. This is attributable to the high per capita consumption of water dictated by modern living standards in today's industrial and service societies [1]. The anthropogenic use of water resources for varying purposes may result in fundamental quantitative and/or qualitative changes in the natural water cycle. This effect has been observed particularly in densely populated areas and in more recent times [2]. Moreover, structural work to control rivers and streams (e.g. as protection against flooding) and drainage of wetlands also have a long-term impact on the physical, morphological, and chemical characteristics of water.

In addition to the increased consumption of water for "traditional" purposes (irrigation, drinking water supplies, hydroelectricity generation), new applications have arisen over the past few decades in the alpine region (e.g. artificial snowmaking in ski resorts), imposing an additional and often significant strain on natural water resources at certain times of the year and in certain locations. It must also be noted that the rich supply of water provided by mountainous regions can be of major importance not only for the population of these regions, but also for residents in lower-lying areas [3].

This chapter focuses on forms of human intervention in the natural water cycle which are typical for the alpine region, and the consequences of such intervention on the quantitative availability of water. Starting with a general overview, the most important quantitative consumption of water, i.e. for electricity generation, is then discussed in detail, including the changes in runoff characteristics and their spatial distribution as well as their effects on other parameters. This is followed by two short sections on artificial snowmaking and irrigation as further examples that typify the consumption of water in the alpine region. The chapter concludes with an overview of the impact of the type of structural intervention in rivers and streams commonly encountered in the Alps, and the effects of changes in land use.

2 Humans and Water in the Alpine Region

2.1 Intervention in the Natural Water Cycle

Surface waters as well as groundwater resources are of paramount importance for the various purposes for which water is consumed. Using the example of Switzerland, Fig. 1 shows the significance of various forms of intervention in the natural water

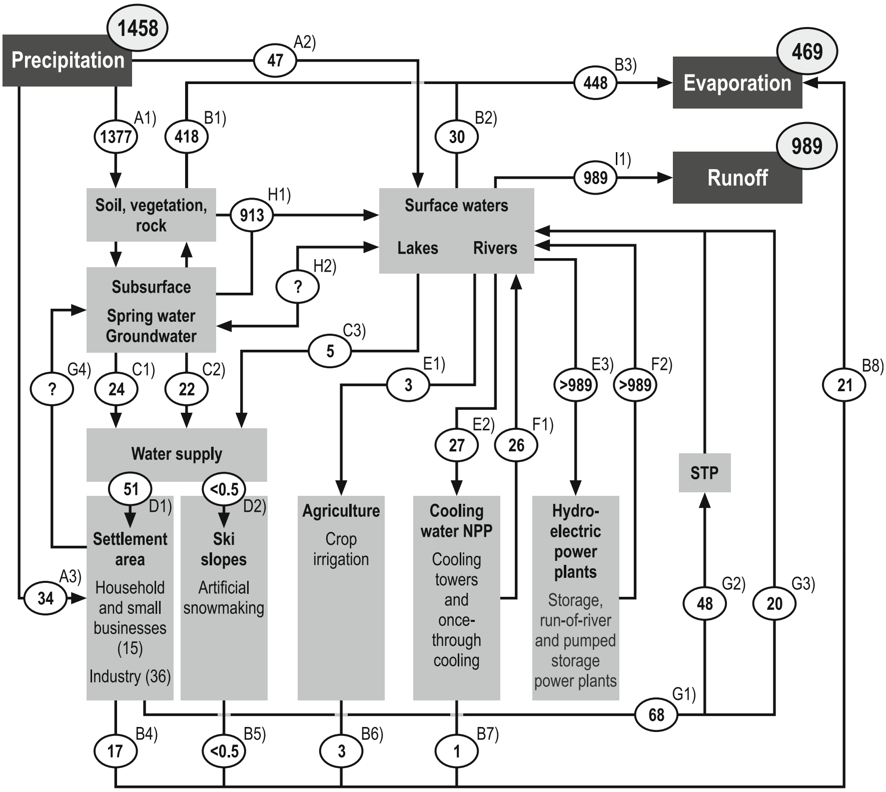

Fig. 1 Water management cycle in Switzerland: all figures quoted in mm/a; 1 mm/a corresponds to 41.3 million m^3/a (acc. to [4] amended, status: early twenty-first century). Key: *A1* Precipitation on natural surfaces (vegetation, soil, rock), [5]; *A2* Precipitation on surface water, [5]; *A3* Precipitation on built-up areas, [5]; *B1* Evaporation from natural surfaces (vegetation, soil, rock), [6]; *B2* Evaporation from surface water, [6]; *B3* Total evaporation from natural surfaces, Calculated; *B4* Natural and additional evaporation from built-up areas, [6]; *B5* Additional evaporation from snow-covered areas, [7, 8]; *B6* Additional evaporation from irrigated farmland, [9]; *B7* Evaporation from cooling towers, Various operators; *B8* Total evaporation from built-up areas and human consumption, Calculated; *C1* Private water supply from groundwater and spring water, [10]; *C2* Public drinking water supply from groundwater and spring water, [11]; *C3* Public drinking water supply from processed lake water, [11]; *D1* Drinking and domestic water supplies to households, small businesses, and industry, [10, 11]; *D2* Drinking and domestic water supplies to snowmaking facilities, [8]; *E1* Water removed from rivers and streams for crop irrigation, [9]; *E2* Water removed from rivers for nuclear power plants (NPP), Various operators; *E3* Water removed and stored for electricity generation by hydroelectric power plants, Estimate; *F1* Cooling water returned to rivers from nuclear power plants, Various operators; *F2* Water returned to rivers from storage power plants, Estimate; *G1* Waste water from industry and built-up areas or meteoric water, Estimate; *G2* Waste water processed by public sewage treatment plants (STP), [12]; *G3* Meteoric water or waste water not treated in private sewage treatment plants, Estimate; *G4* Meteoric water seepage, Unquantifiable; *H1* Natural runoff from soil and underground, Calculated; *H2* Transfer processes between groundwater and surface waters, Unquantifiable; *I1* Runoff from surface waters, [5]

cycle. Note that the order of magnitude of the various forms of water consumption can vary widely in terms of time and space. Likewise, because the volumes withdrawn for specific applications or processes are very difficult to quantify precisely, no numerical information is provided for these cases.

Figure 1 clearly shows that the volume withdrawn by hydroelectric power plants accounts for the lion's share of water consumption. The exact volume used by power plants for generating electricity from rivers and streams cannot be precisely quantified due to lack of data. However, in principle it may be assumed that the entire discharge volume is used once or even several times over for electricity generation.

The production of electrical energy by storage power plants constitutes the dominant form of water use in the alpine region. Since the use of water by these types of power plants can have a significant impact on runoff characteristics and hence on a wide range of other parameters for rivers and streams, the issue is examined in more detail in the following section.

2.2 *Hydro Power as the Dominant use of Water in the Alpine Region*

2.2.1 Principles of Hydro Power

In the alpine region, harnessing water for the generation of electrical energy is an exceptionally important factor, since water in mountainous regions often constitutes the dominant – and even in many cases the only – useful natural resource. Whereas around 20% of the world's electricity is generated from water [13], this percentage can be much higher in areas with a large proportion of mountainous terrain. For example, the volume of hydroelectricity produced in Central European countries increases in direct proportion to the surface area covered by the Alps (Table 1).

As Table 1 illustrates, Switzerland exhibits an extremely high specific annual production of electrical energy from hydro power – by far the largest compared to other countries within Europe. This reflects the high level of water utilization, efficient production facilities, and a high density of power stations: more than 90% of suitable rivers and streams are harnessed in Switzerland for this purpose [13].

With more than 1,200 power stations, the country generates around 57% (=35,000 GWh per year) of electrical energy from hydro power [15]. These power stations are supported by around 1,400 water intake structures [16] and some 200 reservoirs [17]. These artificial storage basins store a total usable volume of almost 4,000 million cubic meters, and are theoretically capable of temporarily retaining around 7% of annual precipitation.

Hydroelectric power plants can be divided into three types according to their storage capability:

Table 1 Gross energy production in selected European countries (Data: [1, 14])

	Surface (1,000 km^2)	of which alpine share (%)	Sectoral ratio of annual gross production of electrical energy (Status: 2005)					Specific production of electrical energy per year (Status: 2005)	
			Hydro power (%)	Wind power (%)	Geothermal (%)	Nuclear (%)	Heat (coal, gas) (%)	Hydro power (MWh km^{-2})	Total (MWh km^{-2})
Austria	84	65.5	58.8	2.1	0	0	39.1	435	741
Switzerland	41	61.5	56.7	0	0	38.1	5.2	793	1,400
Slovenia	20	38.3	24.1	0	0	39.7	36.2	168	699
Italy	301	17.4	14.6	0.8	1.7	0	82.9	141	964
France	544	7.3	10.2	0.2	0	78.3	11.3	103	1,010
Germany	357	3	4.5	4.7	0	26.8	64	74	1,620
Norway	324	–	98.9	0.4	0	0	0.7	418	423
Belgium	33	–	1.9	0.3	0	54.4	43.4	49	2,560
Netherlands	42	–	0.1	2.2	0	3.9	93.8	2	2,320

1. **Run-of-river power plants:** This type of power plant retains the used water and stores it for a short time only. Run-of-river power plants often take the form of power plant chains strung along larger rivers in prealpine and alpine regions, and are a common feature of many alpine rivers, particularly in the middle and lower reaches (e.g. the Rhine). This type of power plant produces a virtually identical volume of electricity over days and weeks.
2. **Storage power plants:** Reservoirs, in many cases with extremely large usable volumes, are a key component of this type of power plant. They store a large proportion of natural discharge and return it to rivers and streams in a monthly and annual equalizing process. These facilities are capable of producing peak load energy to meet peak demand at certain hours of the days.
3. **Pumped-storage power plants:** This is a special type of storage power plant that utilizes a specific volume of water between an upper and lower reservoir. At times of low electrical demand, excess generation capacity from other power plants is used to pump water from the lower reservoir to the higher. At peak load times the water is then released back to the lower reservoir through turbines. During this process, the electrical energy is "refined."

2.2.2 Influence of Power Plant Operations on Runoff Characteristics

The high density of storage power plants in the alpine region affects the natural runoff characteristics of rivers and streams both in the course of the day and over the year. The reduction in and influence of natural runoff frequently constitutes one of the most striking effects on rivers and streams used for power generation, with the degree of influence primarily dependent on the size of the storage facility or system and the way in which the powerhouses are operated. Figure 2 shows the functional principle of a storage power plant as well as the resultant impact on natural runoff characteristics.

As can be seen from Fig. 2 [18], storage power plants collect the water stored in a reservoir by means of inflows (within the catchment area) and diversions (from other catchment areas). Often this leaves only a minimal flow of water in the so-called residual flow reach below the withdrawal sites. In some cases, water from other hydrological basins can even be diverted over main watersheds. For example, more than 50 million cubic meters of water per year is diverted from the River Unteralpreuss (Rhine basin) to Lago Ritom (Po basin) [19].

When demand for energy is high, a storage power plant releases water to turbines in a powerhouse which in many cases is not situated on the dammed river. The water is led through pressure pipelines that do not follow the natural water course, as a result of which little or no water is run off in the residual flow reach below the reservoirs. By contrast, the runoff is higher than would naturally occur after the point of reflux.

Because most reservoirs in the alpine region are designed as annual storage basins, the annual runoff volume following the convergence of reflux and residual

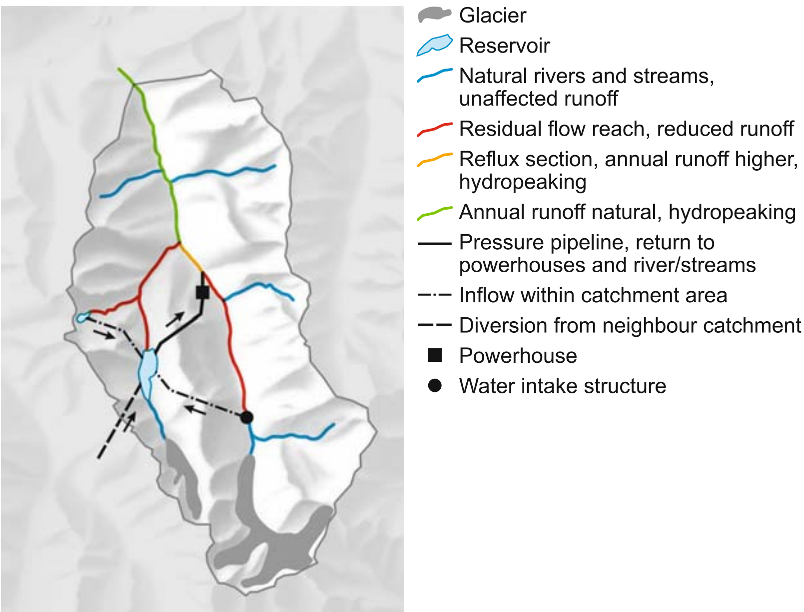

Fig. 2 Functional principle of a storage power plant and its impact on runoff characteristics (acc. to [18] modified)

flow reach is roughly equivalent to the natural runoff volumes. However, since hydroelectricity production is dictated not only by seasonal demand but also by short-energy requirements, not only changes in monthly runoff but also major daily fluctuations in runoff can be expected ("hydropeaking"). This exerts an influence on natural runoff characteristics on varying time scales.

Change in Monthly Runoff Volumes

Influence on monthly runoff volumes or the annual runoff regime can take two basic forms, which differ depending on the type and intensity of hydroelectricity consumption (Fig. 3).

1. **First basic type (A and B): Shift in seasonal runoff volumes.** The high volumes of runoff which have accumulated during the summer due to water storage in reservoirs are shifted to the winter half-year, when demand for electricity is higher. This is manifested in higher winter runoff and reduced summer runoff, resulting in a balanced runoff regime (Type A). This influence can be somewhat mitigated by larger natural lakes (e.g. on the alpine periphery) (Type B).
2. **Second basic type (C and D): General reduction in runoff volumes.** An even more pronounced influence on the runoff regime occurs when water is withdrawn from a river or stream and diverted to another catchment area. Depending

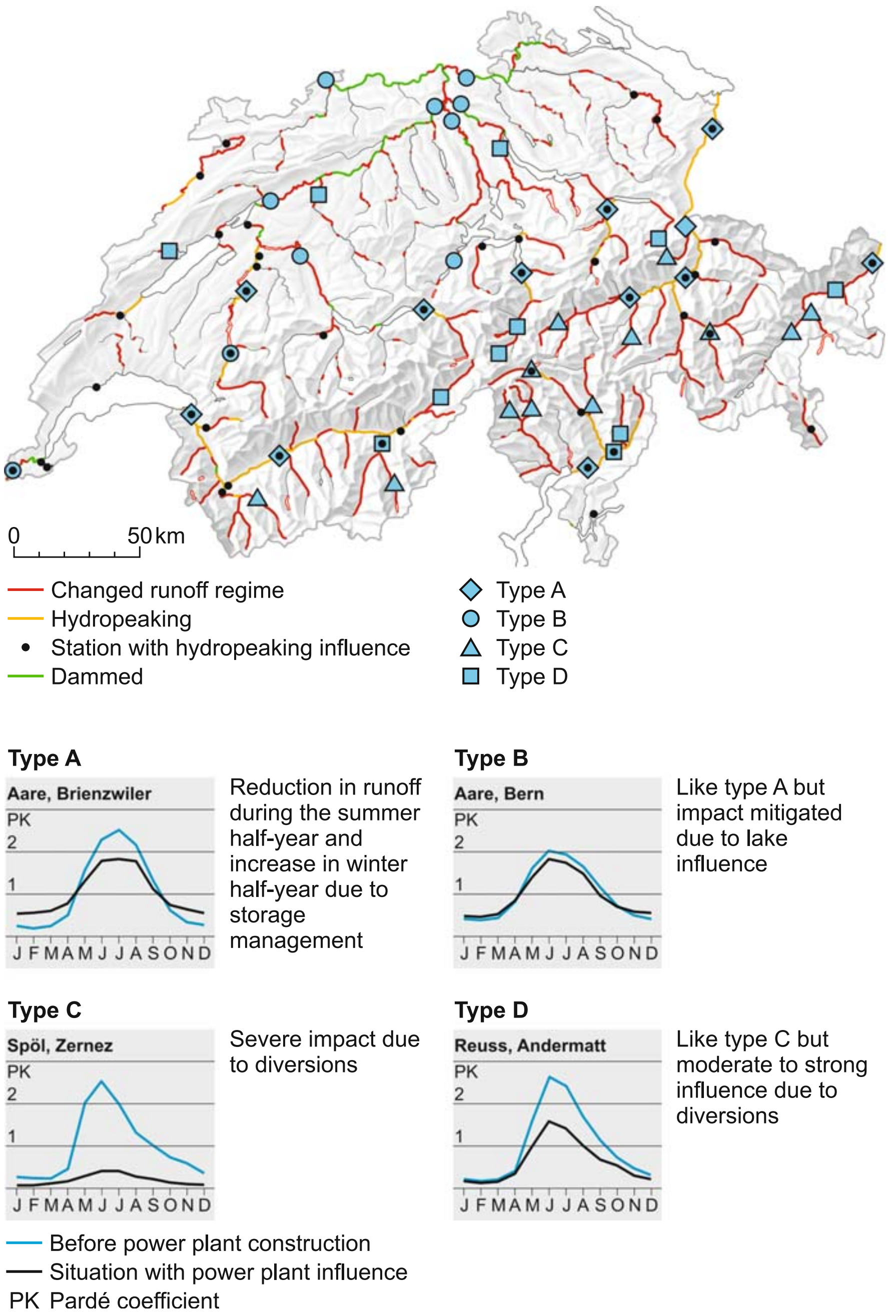

Fig. 3 Different types of runoff regime influenced by power plant operations, and their spatial distribution in Switzerland (acc. to [19, 20] amended and supplemented)

on the volumes withdrawn, this may even result in the widespread drying up of residual flow reaches. Although the minimal residual flow volume is legally regulated in Switzerland, at present extensive sections of the affected waters still require remediation [21, 22].

In addition to measurement stations and reaches of river where affected runoff regimes occur, Fig. 3 also shows measurement stations and reaches of river in which hydropeaking characteristics have been identified [19, 20]. It also shows river reaches which have been dammed for use by run-of-river power plants. The presentation is limited to larger rivers in Switzerland.

The quantity-related changes in mean monthly runoff are also evident in the example of the catchment area of the Rhone down to Porte du Scex, which is heavily utilized by storage power plants (Fig. 4a). The catchment covers a surface area of 5,244 km^2. The average altitude is 2,130 m ASL and the ice cover is 14.3%. Over the past 100 or so years, the useful capacity of reservoirs has been expanded to more than 1,200 million cubic meters by constructing around 50 dams (Fig. 4b). Here the quantum growth in hydroelectricity between 1950 and 1970, which is typical of the alpine region, can be clearly identified. Thanks to the capacity of reservoirs in the catchment area, around one-fifth of the annual runoff volume can be temporarily stored.

The five-year Pardé coefficients (Fig. 4a) exemplify the way in which the typical "glacionival" runoff regime [24] has developed a balanced, unnatural pattern. Now, because of the influence exerted by power plants, more water is discharged in the winter half-year and less in the summer half-year than in the decades where such influence was absent.

Change in Daily Runoff Volumes

The influence exerted by power plants on runoff characteristics is also clearly illustrated by the daily averages. Over the past 100 years, the number of days with maximum and minimum mean daily runoff on the River Rhone (Porte du Scex) was drastically reduced (Fig. 4c and d). While the runoff limit of 400 $m^3\ s^{-1}$ was exceeded on an average of 45 days per year between 1905 and 1954, between 1955 and 2004 this only occurred on an average of 18 days per year. The change is even more pronounced in small runoffs: in the phase with no influence from power plants, the daily mean runoff was less than 50 $m^3\ s^{-1}$ on an average of 54 days per year, whereas in the phase influenced by power plants this only occurred on an average of 2 days per year.

While the daily averages in the phase influenced by power plants exhibit significantly less variability, a very different daily runoff pattern is often observed: during the day, short-term surges and downsurges (hydropeaking) can be observed over wide sections of the main rivers. This artificially induced high water impacts

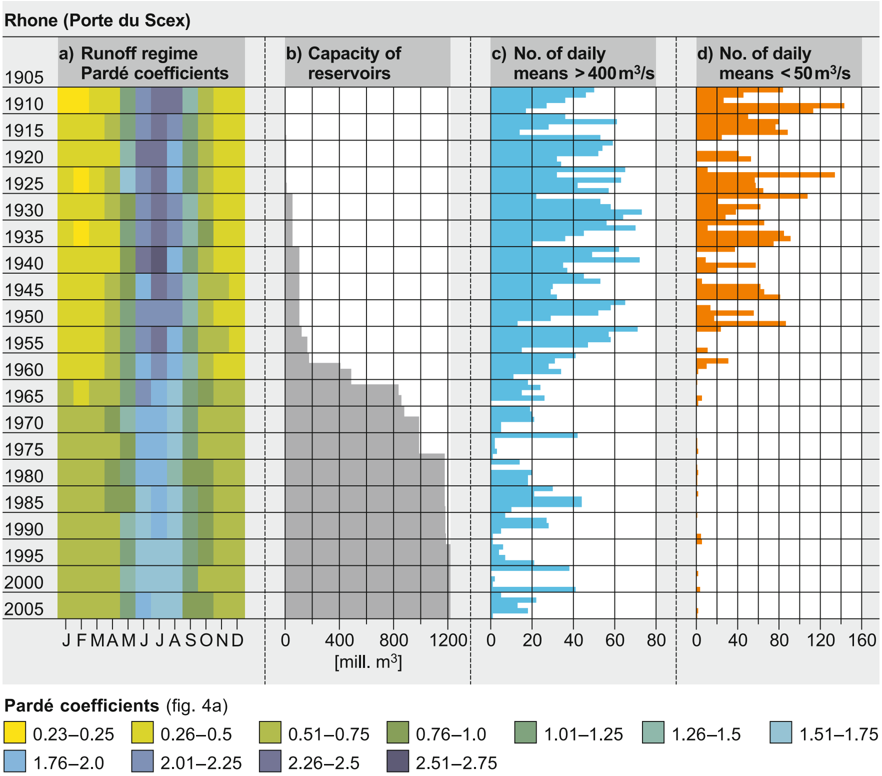

Fig. 4 Case study Rhone (Porte du Scex, 1905–2004): usable volume in reservoirs in the catchment area and associated changes in runoff characteristics (Data: [17, 23])

medium-sized rivers up to 40 km below the point of reflux [25]. Typically, an extremely rapid rise is observed around mid-day and an equally rapid fall towards evening, whereby a pronounced weekly cycle of this artificial diurnal variation is identifiable, particularly in winter. This affects working days much more strongly than weekends. Moreover, the regular and frequent changes in runoff usually occur at a faster speed than is the case for natural flood events [26].

The order of magnitude of runoff changes can be described using the surge-downsurge ratio (ratio between the highest and lowest runoff). At individual stations on Swiss waters, this can amount to more than 20:1, while the literature lists a range of 3:1–5:1 from which significant environmentally relevant effects can be expected [23]. Nevertheless, to date most countries have not defined any generally recognized and binding thresholds.

It is assumed that approximately 25% of the 500 or so power plants in Switzerland (primarily of the storage power type) trigger hydropeaking in runoff [26]. Although no new power plants have been built in Switzerland since the 1970s, there has been an increase in various hydropeaking indicators at several stations, although several have also seen a decrease [23]. The identified differences can be explained by the increased construction of powerhouses, enlargement of storage capacity or modified operating concepts [27].

Short-Term Influence of Floods

Although not designed to retain discharge from floods, thanks to their retaining effect most reservoirs attached to storage power plants in the alpine region can make a valuable contribution to reducing flood risk. Exceptionally severe high water occurred in the Rhone catchment area in 1987, 1993 and 2000, resulting in massive flooding in some locations, accompanied by severe damage to property [28]. An analysis of these events provides a clear illustration of the influence exerted by reservoirs in cases of extreme flooding (Table 2).

The floods of August 1987, September 1993, and October 2000 occurred at various points in the annual reservoir filling process. Since reservoir volumes are at their lowest in spring and highest in autumn, the time at which flood-inducing precipitation occurs is an important factor in determining their ability to retain runoff [30]. In the Rhone catchment area, around 10% of reservoir capacity was free at the beginning of the flood events in August 1987 and October 2000, which ultimately enabled a precipitation volume of 60% to be retained on each occasion over the entirety of the event (Table 2). Conversely, due to the larger volumes stored in these reservoirs in September 1993, only 50% of the water discharged could be retained.

2.2.3 Extent and Spatial Distribution of Changed Runoff Characteristics

The large number of water diversion points, reservoirs, and powerhouses exert an influence on the runoff characteristics along extensive sections of Switzerland's rivers and streams [19, 31]. However, the extent of the influence varies widely not only according to region but also depending on the size of the waterway (cf. Fig. 5).

An evaluation of the basic data, which for Switzerland are on a scale of 1:200,000 [16, 19], produces the following picture: the runoff characteristics of more than 70% of Switzerland's main waterways (excluding lakes) are influenced by anthropogenic use. More than half of these (accounting for 40% of all Swiss

Table 2 Rhone (Porte du Scex): Comparison of retention impact of reservoirs in the catchment area during extreme flooding events in 1987, 1993, and 2000 [29]

	August 1987	September 1993	October 2000
Level at start of event (%)	89	95	90
Additional inflow (1,000 m^3)	69,600	64,900	66,800
Stored volume (1,000 m^3)	41,200	31,200	40,000
Stored volume (%)	59	48	60

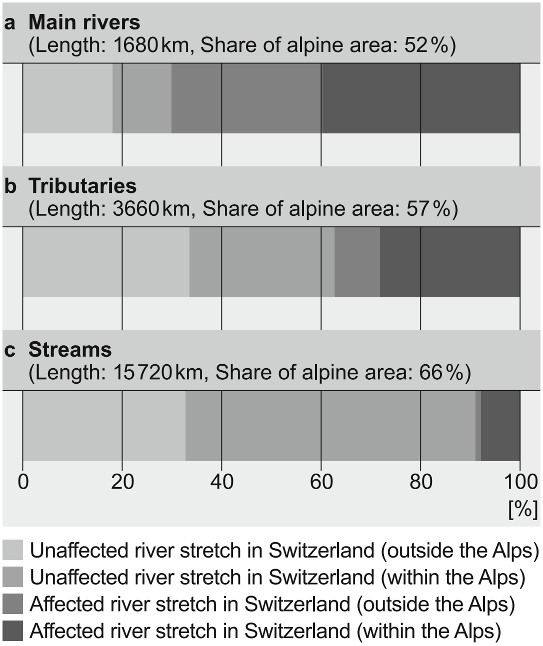

Fig. 5 Affected runoff characteristics of Swiss waters. By river stretch, river size, and region. (Data: [16, 19])

waterways) are in the alpine region. Of the sections of main Swiss waterways which are unaffected, a good 10% are situated inside the Alps and close to 20% outside the Alps. Affected runoff characteristics are those which exhibit seasonal changes in the runoff regime, hydropeaking characteristics, or a marked reduction in runoff volume (Fig. 5a).

In principle, changes in runoff characteristics can be said to occur primarily in Switzerland's main waterways. In the case of tributaries, the ratio of unaffected sections is more than 60% (Fig. 5b), and more than 90% for larger streams (Fig. 5c). This points to the fact that rivers and streams of these dimensions are no longer economically viable for use in electricity generation. However, many unaffected sections of larger streams are situated above the first water diversion point.

It should be noted that no additional aspects were taken into consideration in this compilation other than the influence on runoff characteristics. Any evaluation of morphological, biological, physical, or chemical parameters would produce a very different pattern.

2.2.4 Impact of Changed Runoff Characteristics

Reservoirs, which are virtually exclusively situated in the alpine region, generally store water from glaciated catchments. In low-precipitation years, snow and ice melt from glaciers compensates for runoff deficits, and in so doing permanently ensures a minimum volume of water in rivers and streams. In addition, however, glaciers play a key role in releasing sediment. Both these factors exert a significant influence on the morphology of alpine waters and, coupled with the temperature, can dictate the composition of biocoenosis of aquatic animals and plants [32–34].

They also have an impact on runoff characteristics as well as many physical parameters of rivers and streams. For instance, hydraulic attributes (width of water level, flow velocity, depth, tractive force, or shear stress) and hence the bed-load regime and temperature can alter the natural state.

Changes in the runoff regime can also have an aesthetic impact on the landscape or influence other forms of use (e.g. irrigation, drinking water supplies) due to the absence or reduction in volumes of water [35].

The following two examples illustrate the effects of changed runoff volumes:

Effects on the Solid Matter Regime

By trapping the sediment from used alpine waters, reservoirs reduce the transport of suspended load to residual flow reaches. In these sections with a reduced channel flow, the tractive force and shear stress is drastically reduced. This additionally reduces the bed load transport, which may then result in solid matter originating from unaffected tributary streams remaining in the main channel, thus significantly increasing the debris-flow hazard for episodic high water discharge [36].

The reduced flow velocity can also change the substrate composition in the channel and the bulk density of the river bed. This often gives rise to clogging, which impedes the exchange between running water and groundwater [37]. Hydropeaking can exacerbate this process. Small-scale upward and downward fluctuations linked to frequently changing currents contribute to the efficient layering of fine particles and hence fill the interstitial system [38, 39].

Effects on the Temperature Regime

As the studies of [40] show, the annual temperature regime of alpine waters can also change under the influence of power plant operations. At points of reflux in the Rhone, the average temperature in winter is now approximately 2°C higher and in summer around 1°C lower than under natural conditions. The cooler summer temperature is due to the fact that relatively cool deepwater from reservoirs is passed through turbines at this time of year. But the reduction is within the range of naturally occurring annual fluctuations. The higher temperature in winter is attributable to greater volumes of runoff which stop the water from cooling down.

Yet, due to the reduced volume of runoff and flow velocities, a much higher temperature – as much as several degrees – has been observed in residual flow reaches during the summer.

2.3 Further Examples of Water Consumption in the Alpine Region

In addition to being used for water supplies, water in alpine tourist regions plays an important role in other applications such as irrigation, artificial snowmaking, leisure, and as a landscape feature [41]. It must be noted here that specific applications are concentrated during the main tourist season. Particularly during high season and at weekends the tourist population often far exceeds the local population, which normally results in even higher consumption of water for specific purposes. These temporary additional strains on supplies can be so severe that they give rise to water shortages [42, 43]. The situation is particularly unfavorable during the winter months, since at this time of year the spring discharges, which are often of paramount importance in alpine regions, exhibit minimal runoff [35].

The following section briefly discusses two types of water use which are typical of the alpine region.

2.3.1 Artificial Snowmaking

Since the end of the 1970s, many regions of the Alps have been using water during the winter months to make artificial snow for ski slopes. Although the volume of water used for this purpose is often stored in special-purpose reservoirs during the entire summer season, drinking water supplies are often also resorted to. The extent of this additional consumption can be sizeable, particularly at the local level: today, the volume of water used in individual regions can amount to 20–35% of the drinking water consumed per region and season [8].

In 2006 around 35% of all ski slopes in the Alps – or roughly 35,000 hectares – were covered with artificial snow. But a comparison with other European countries shows major differences (Fig. 6). If one assumes that 2,000 m^3 of water per hectare need to be used for artificial snowmaking [8], this results in an annual water requirement of 70 million cubic meters for the alpine region as a whole. The estimated volume for Switzerland is around 8.4 million cubic meters.

In addition to the wide range of effects which artificial snow has on the environment (e.g. [46–49]), the loss of evaporation and sublimation resulting from snow cover or snowmaking is of particular relevance for the water balance of alpine catchment areas. The extent of this loss is still unclear. The data range from 10% to 30% of the volume of water used for snowmaking [7, 8].

Even if one assumes that artificial snowmaking equipment will be used on many more ski slopes in the alpine region in the future, the influence of this application on the natural water cycle – at least from a large-scale viewpoint – is of secondary importance. However, intensive snowmaking is likely to exacerbate local shortages and conflicts related to water supplies and use.

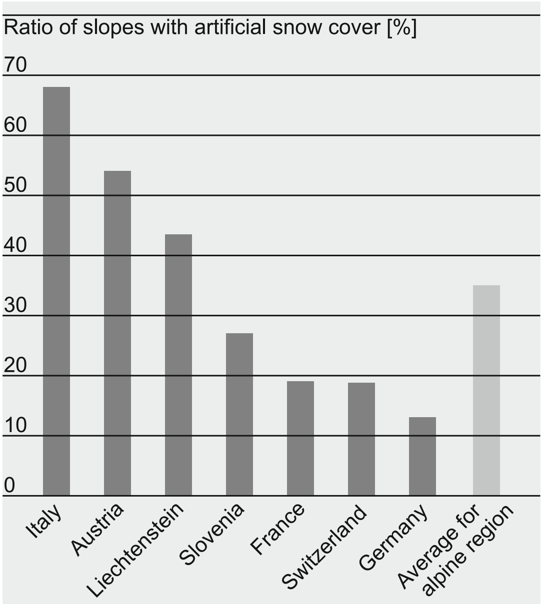

Fig. 6 Ratio of slopes in the alpine area with snowmaking facilities (Status 2006 (Slovenia 2004); Data: [44, 45])

2.3.2 Irrigation

There is a long tradition of irrigation for agricultural purposes in the alpine region, particularly in the dryer central valleys (Valais, Aosta Valley South Tyrol), and irrigation was a common practice in the late Middle Ages and even as far back as Roman times. Besides transporting humidity, running water used primarily for irrigation causes other effects: in spring the ground is warmed and the irrigated surfaces are fertilized by the nutrients released in the water [50].

In 2007, 144 million cubic meters of water was used in Switzerland to irrigate around 43,000 hectares on a regular basis and 12,000 hectares on an occasional basis, with the inner-alpine dry valley of the Valais accounting for more than half of the irrigated surface area [9]. The water used for this purpose is taken primarily from water courses, particularly during dry seasons or climate periods, and used directly without any interim storage.

Because of the effects of climate change, it must be assumed that the irrigated surface area will extend in future to 150,000 hectares, with the associated growth in demand for water [51].

2.4 Structural Control and Correction of Water Courses

Over the centuries, river valleys and plains in the alpine region and the foothills have been affected by regular flooding. For a long time, large areas of these natural floodplains were uninhabitable or unfit for farming. Added to this, the mountainous regions were exposed to the risk of debris flow and high water from mountain torrents.

In the nineteenth century, population growth resulted in an increased demand for housing, particularly in the lower regions. At the same time there was an increase in runoff and bed load caused by the growing depletion of forests and triggered by the prevailing climate phases. For these reasons, and thanks to the enormous progress made in hydraulics, a number of river corrections were carried out from the end of the eighteenth century onwards, initially in the valleys and subsequently in the upper reaches of rivers and streams, accompanied later by structural controls on mountain torrents [50, 52]. In the course of the nineteenth and twentieth centuries, most rivers in Switzerland were structurally corrected and many peripheral alpine lakes were regulated. Valley floors were drained and water courses were dammed and straightened for a variety of reasons [53]. In addition to improving floor protection and gaining additional land for agricultural purposes, these measures also reduced the risk of epidemic and optimized the use of waters for transportation purposes.

Apart from the direct impact of water use on natural runoff characteristics, structural changes on surface waters constitute the most striking visual change in the natural system.

Since 1998 Switzerland has conducted a nationwide survey on the ecomorphological state of its waters, which records and evaluates all the structural characteristics of each water course [54]. A projection for the whole of Switzerland is predicated on the fact that around 25% of the sections of all water courses (=15,800 km) are in a poor ecomorphological state and 50% are still in a natural state. However there is a very close correlation between the extent of the impact and the altitude: below 600 m ASL, 50% of the water courses exhibit insufficient structural variety, while above 2,000 m the ratio is only 2%. However, the data collected is not yet reliable, particularly for higher locations in the alpine region.

In conjunction with the morphological influence, transversal and lateral structures exert a significant impact. Switzerland's hydrographic system has approximately 88,000 structural interventions of this type at heights of >50 cm, which corresponds to more than one major obstacle per kilometer of river. These structures have the potential, for example, to cause massive changes in fish migration, which are all the more crucial bearing in mind the fact that even one large sectoral structure renders an entire river section inaccessible to fish.

Unlike the evaluation of sections with affected runoff characteristics, these parameters show that small waterways are more strongly influenced (1.6 obstacles per kilometer) than larger water courses (0.6 obstacles per kilometer). However, settlement areas exert the largest influence (2.5 obstacles per kilometer), which is a logical result of the higher need for flood protection and the related level of structural controls [54].

The effects of longitudinal and transversal structures for the purpose of correcting waterways, however, has further implications for various key characteristics of waterways. Table 3 provides an overview ([55] modified).

Table 3 Effects of straightening and structural controls on selected river/stream parameters (acc. to [55], amended)

	Unregulated, braided river	Controlled, straightened river; without weirs	Controlled, straightened river; with weirs
Average flow velocity	Low	High	Medium
Overbank area in times of flood	Large	Small	Small
Structural and current diversity	Large	Small	Rel. small
Groundwater level	High	Rel. low	Rel. high
Ease of passage for fish	Unaffected	Unaffected	Severely affected

2.5 *Changes in Land Use*

Changes in land use triggered by anthropogenic as well as climate change can also have an indirect effect on quantitative runoff characteristics in alpine catchment areas.

The extent of the impact of land-use changes on runoff processes is a subject of controversy [56, 57]. In principle, however, it is safe to say that the type of vegetation, its spatial distribution and in particular the soil depth or composition can exert a strong influence on the runoff regime in an (alpine) catchment area.

Thus, any reduction in vegetation cover causes a reduction in water retention and evaporation, which ultimately increases surface runoff and the runoff concentration times [58]. Human-induced soil sealing and soil compaction has the same effect (Fig. 7).

Nevertheless it should be noted that, in the configuration shown in Fig. 7, the runoff processes are not dictated solely by land use or the type of vegetation. For example, [59] states that, while forest areas can reduce the flood disposition of a catchment, the extent of the retaining effect is primarily determined by the characteristics of the forest soil. Added to this, forests also always consume large volumes of water since they exhibit higher evapotranspiration than other types of vegetation.

The gradual spread of sealed surface areas also contributes to the reduction in evaporation and to seepage of precipitation water, resulting in the more rapid discharge of a higher proportion of precipitation. These effects are particularly noticeable in the case of mean and high water, but less so in the case of extreme high water events [55]. Reduced seepage can severely reduce water storage in the soil, leading to lower volumes of low water runoff in dry phases and reducing evaporation and the recharge of groundwater [60]. This effect can also be observed in water courses in catchment areas where soil reclamation measures are being carried out.

“Water Storage and Water Balance” in [61] cited the ability of glaciers to deliver runoff as an important influencing factor on runoff characteristics in alpine

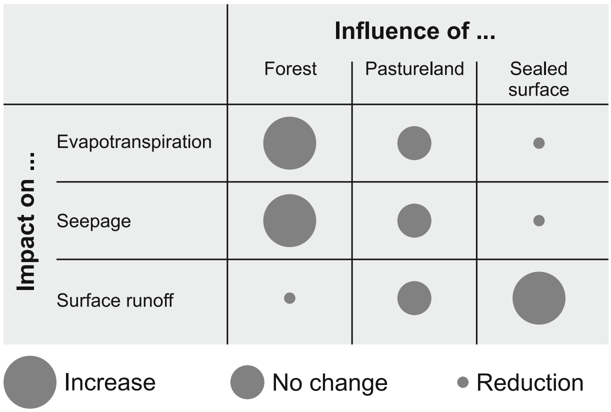

Fig. 7 Potential impact of land-use changes on selected water cycle elements and runoff processes (acc. to [55], amended)

catchment areas. In future, the global rise in temperature due to climate change, and the associated retreat of glaciers, will lead to a reduction in ice melt, which in turn will alter the present runoff pattern in alpine glaciated catchments: in summer and autumn, runoff volumes will be lower than they have been to date, whereas in spring they will be significantly higher and in winter slightly higher (e.g. [62]), i.e. the result will be a balanced runoff regime. This effect far outweighs the impact of human-induced land-use changes on monthly runoff volumes in heavily glaciated catchments.

3 Conclusion

Since water in alpine regions often constitutes the only natural resource, and because the Alps are characterized by much higher water supplies than their foothills, water use is an extremely important factor in these regions. In addition to providing supplies of drinking water, water in the alpine region is typically used to generate hydroelectricity, to irrigate crops and for artificial snowmaking. In particular, the dominant use of water for hydroelectricity generation in the Alps can significantly change quantitative runoff characteristics on various time scales and on a supra-regional basis.

Figure 8 shows the human-induced effects (darkgrey) that can affect the quantitative runoff characteristics of alpine waters over time and space. For the sake of comparison, this figure also incorporates natural effects (lightgrey).

It is evident from this that natural as well as human-induced effects can produce similar runoff patterns. Thus, for example, local short-term changes in runoff can be triggered by heavy precipitation events (flooding) as well as by the feedback of turbine-processed water from power plants (hydropeaking). Usually, however,

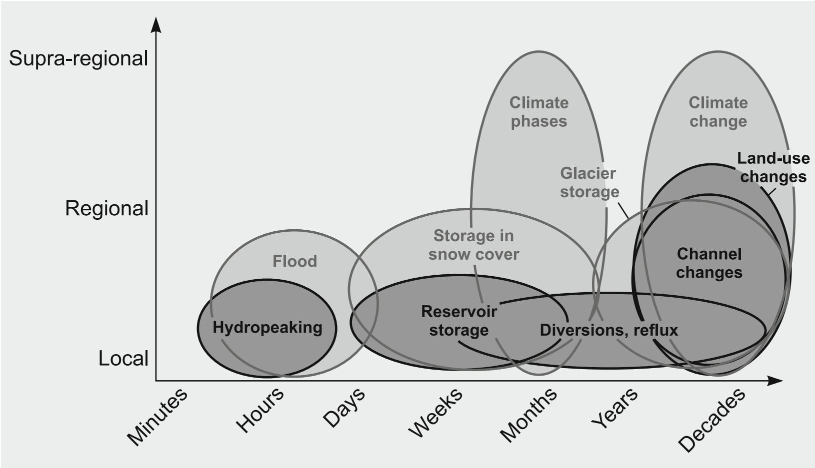

Fig. 8 Overview of natural and anthropogenic influences on runoff characteristics

human-induced runoff patterns are more pronounced than the corresponding natural runoffs: for instance, hydropeaking can exhibit a much shorter rise time and occur at other times of the year than natural flooding. Human-induced changes in quantitative runoff characteristics therefore frequently impact ecosystems as well as many of the morphological, physical, and chemical parameters of alpine waters.

Since the effects of climate change will probably result in changes in runoff pattern, and a growing demand for water for various applications must also be assumed, the strain on natural water supplies in alpine regions will also be concentrated at particular times and be significantly stronger at the local level. This could result in a shortage of water resources at certain times and in certain locations, which – given the important role that mountainous regions play as water towers for the foothills – may also impact the population in surrounding regions.

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Part II
Biogeochemistry and Pollution of Alpine Waters

Water Chemistry of Swiss Alpine Rivers

Jürg Zobrist

Abstract Average concentrations of dissolved nutrients (NO_3, DRP, K) in the large alpine rivers Rhine, Rhône, Ticino, and Inn, and in small alpine streams and glacier streams, are low compared to those in midland rivers. Concentrations of NO_3 in the large rivers clearly exceed background concentrations. In spite of limited anthropogenic activities in alpine catchments, DRP concentrations in large rivers exhibited a downward trend over the last 30 years. Time series of NO_3 concentrations were first increasing and then leveled off. Export coefficients of NO_3 and DRP in alpine streams fall in the range of those estimated for nonagricultural lands and forests on the Swiss Plateau.

The chemical weathering rate of rock-forming minerals in alpine catchments is about 165 ± 45 g m^{-2} y^{-1}, corresponding to an ablation rate of about 0.06 mm y^{-1}. Rates are dominated by the reaction of carbonate-containing rocks with CO_2 and the dissolution of anhydrite, whereas the weathering of silicate minerals contributes little. Total chemical weathering rates are in the same range as the export rate of fine sediments, as part of physical weathering products. In this respect, alpine rivers differ distinctly from lowland running waters. Long-term observations also revealed small changes in concentrations and loads of geochemical constituents. An increase in water temperature may be one driver for these changes, although other factors also play a role.

Keywords Export coefficients, Nutrients, Temporal trend, Water quality, Weathering

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Contents

Abbreviations

Ca	Calcium
CO_2	Carbon dioxide
DRP	Dissolved reactive phosphorous
H_4SiO_4	Silicic acid
K	Potassium
Mg	Magnesium
NO_3	Nitrate
P	Phosphorous
SS	Suspended solids
TP	Total phosphorous

1 Introduction

1.1 Specific Characters of Alpine Rivers

Alpine rivers in Middle Europe represent a special aquatic ecosystem and their characters differ greatly from those of lowland rivers (see [1], this volume). Alpine rivers have their sources in mountain regions where the land is covered with snow for several months of the year and where glaciers can exist in high altitude areas. As a result, water discharge decreases to a minimum in winter and a

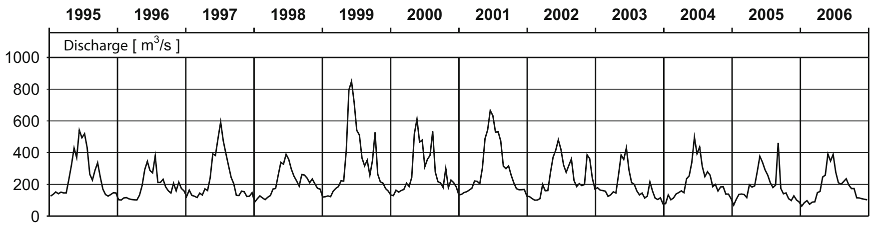

Fig. 1 Seasonal pattern of water discharge in the alpine Rhine (Diepoldsau) from 1995 to 2006 [3]

maximum discharge occurs during snowmelt in early summer (Fig. 1). During this period, much sand, silt and clay, as well as gravel, are transported downstream by turbulent flows, i.e. the river water is turbid, and exhibits a deep grey color and a high concentration of suspended solids (SS). In winter, alpine running waters mostly have clear waters and a laminar flow.

A temperate-humid climatic zone surrounds the Alps. Due to orographic rains, yearly precipitation rates may increase regionally to over 2 m y^{-1}, whereby summer precipitation rates can exceed those in winter (for details see Figs. 6 and 8, [2], this volume). The runoff from alpine catchments generally varies between 0.9 and 1.4 m y^{-1}, i.e. in the range of 28–45 L s^{-1} km^{-2} ([2, 3], this volume), depending on region and year. This high discharge rate illustrates a typical property of alpine running waters. Rates are considerably greater than in lowland middle European rivers such as the Elbe, Seine (0.20 $\pm$ 0.05 m y^{-1}) and also water inputs into the sea by the rivers Danube, Rhine, and Rhône (0.25–0.56 m y^{-1}) [4].

Presently, a large part of the river water in alpine regions is used for producing hydroelectric power. This abstraction of water may drastically change the flow regime in some segments of rivers. In addition, hydropeaking (rapid changes in water discharge due to temporary running of hydroelectric power stations) may strongly impair the ecological condition of these running waters [5]. Wehren ([6], this volume) provides more detail regarding disturbances due to human interventions.

1.2 Processes Regulating Water Composition

Figure 2 shows processes and factors that regulate water quality.

The condition or state of a natural water body can be characterized by a set of measurable physical, chemical, and biological parameters and a description of observed phenomena. The condition results from the various inputs of diffuse and point sources from natural and anthropogenic origins. These inputs may be transformed from the place of input to the place of measurement depending on the character of the catchment and parameter. Water quality always reflects

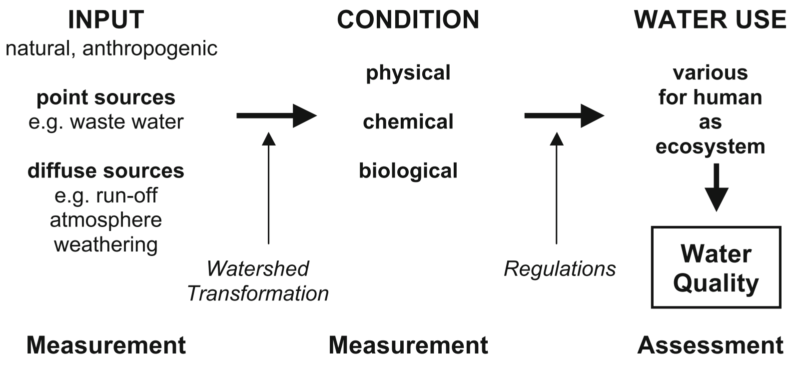

Fig. 2 Schematic representation of the relationship between input into a river, condition (state), water use, and water quality

an interpretation of data gained by observing the water condition. This assessment must be accomplished according to the goal of the water use and regulations, such as the EU Water Framework Directive [7] or the Swiss Water Protection Ordinance [8].

The water composition of alpine rivers is strongly influenced by natural diffuse inputs originating from rock weathering. Weathering processes are defined as the alteration of rocks in the top crust of the earth; it is mainly an interaction of water with rocks [9, 10]. Water can act through physical and chemical processes.

In physical weathering, the rock is broken apart by frost heaving, glacial plucking, and abrasion in running waters. The products of these processes are solid particles (rock debris) exhibiting various sizes (boulder, gravel, sand, and silt). In alpine regions, the subsoil consists mainly of these weathering products. A percentage of the smaller particles are flushed downhill by surface runoff, thus representing the SS in running waters. Therefore, alpine rivers exhibit a large load of SS during high water discharge caused by snowmelt or heavy rainfall. Physical weathering also increases the surface area of the rock and debris exposed to chemical weathering.

Chemical weathering describes the interactions of rock-forming minerals with water and its solutes (protons, carbonic acid, complexing agents) [9]. For the most abundant rocks, calcareous rocks and silicates, such as granitoid rocks, chemical weathering can be considered as acid–base reactions in which the rock-forming minerals represent the base and the dissolved reactants act as the acid, especially dissolved CO_2. Other minerals, such as gypsum, simply undergo dissolution reactions in water (Table 1).

Biological weathering involves the disintegration of rock and its minerals by the chemical and physical actions of living organisms, for example plant roots or bacteria, which can dissolve minerals by oxidation or reduction processes [11].

The rates of chemical weathering reactions increase with temperature as all chemical processes. This effect has been confirmed in laboratory experiments and field studies involving silicate rocks [12]. Weathering reactions occur at mineral

Table 1 Important weathering reactions in order of ease of chemical weathering and solubility, which goes along with the reaction rate of the mineral dissolution, except for bacterial mediated pyrite oxidation [9, 10]

Rock-forming mineral	Equation			Solubility
Halite (Sylvine)	NaCl (KCl)	$\rightleftharpoons$	$Na^+ (K^+) + Cl$	High
Gypsum	$CaSO_4\ 2H_2O$	$\rightleftharpoons$	$Ca^{2+} + SO_4^{2-} + 2H_2O$	High
Calcite	$CaCO_3 + H_2CO_3$	$\rightleftharpoons$	$Ca^{2+} + 2HCO_3^-$	Intermediate
Dolomite	$CaMg(CO_3)_2 + 2H_2CO_3$	$\rightleftharpoons$	$Ca^{2+} + Mg^{2+} + 4HCO_3^-$	Intermediate
Silicates	Primary minerals + H_2CO_3	$\rightarrow$	Base cations + $H_4SiO_4 + HCO_3^- + Al(OH)_3$ + secondary minerals	Slight
Quartz	$SiO_2 + 2H_2O$	$\rightleftharpoons$	H_4SiO_4	Slight
Apatite	$Ca_{10}(PO_4)_6(OH)_2 + 6H_2O$	$\rightleftharpoons$	$4\ Ca_2(HPO_4)(OH)_2 + 2Ca^{2+} + 2HPO_4^{2-}$	Low
Pyrite	$FeS_2 + 3.75O_2 + 2.5H_2O$	$\rightarrow$	$Fe(OH)_3 + 2SO_4^{2-} + 2H^+$	

Notes:
CO_2 consumed by weathering reactions represents a sink for atmospheric CO_2
Dissolution of pyrite can also be mediated by bacteria. Weathering reaction listed for silicates is elementary only

surfaces; therefore, the overall rate relies on the size of the wetted surface of the mineral particle and the water flow regime. Hence, comparisons between mineral dissolution rates determined in the laboratory and in the field reveal large discrepancies with orders of magnitude lower rates in the field [9, 13, 14]. In addition, environmental factors such as the mineral supply from the breakdown of rocks [15] or runoff [16, 17] can influence weathering rates occurring in natural systems. Factors that govern weathering rates can also be described by two kinds of weathering regime [18]. In a transport-limited regime, physical erosion is low and produces fewer new reactive surfaces of minerals. In addition, it leads to an excess of soil that depresses the chemical weathering of bedrock. In a weathering-limited regime, high physical erosion restricts soil formation and continuously creates new reactive surfaces that can be weathered chemically. Weathering rates of carbonates and silicates also depend on the partial pressure of CO_2 at the location where reactions proceed in soil and aquifers. The solubility of calcite and dolomite is inversely proportional to temperatures found in natural aquatic environments [19]. All these processes and factors must be considered when assessing whether climate effects change weathering rates and its implication regarding the regulation of atmospheric CO_2 [12, 15, 18, 19].

In alpine rivers, dilution will govern the impairment from pollution loads by point or diffuse sources such as treated wastewaters or runoff from agricultural lands. This dilution effect decreases gradually when rivers enter the lower altitude range of a watershed.

1.3 Data

In the field of water protection, small and medium-sized alpine running waters are rarely monitored, since their impairment by pollution is considered small. Therefore, the database of nutrients for such rivers relies on ecological research projects [20–23]. The National Long-Term Surveillance of Swiss Rivers (NADUF) [23] monitors the larger alpine rivers Rhine, Rhône, Ticino, and Inn, and yields data on nutrients and geochemical constituents as well. To highlight the specific chemical condition of alpine rivers, relevant data, including the Thur River for comparison, are included in the following discussion. The Thur flows through a part of the eastern Swiss Plateau (Swiss Midlands) where major intensive agricultural activities occur. The Thur has its source in the alpine region of the Säntis massif.

Rivers listed in Table 2 and shown in Fig. 3 incorporate a broad range of characteristics, including water discharge, catchment area, land cover, land use, and the number of people living in the catchment. Land use and population reflect the potential pollution intensity, whereas the percentage of intensively used agricultural land in the catchment or the number of inhabitants per unit water discharge are major pollution stress factors. They also represent important diffuse and point sources of pollution.

Table 2 Character of alpine catchments for which chemical data are discussed

River station	Water discharge $m^3 s^{-1}$ mean Q	Basin area km^2	% Barren	% Forest	% ext. used	% int. used	% settl.	Inhab. per mean Q	Basin altitude m.a.s.l. mean	References
Six glacier streams	0.6–4	20–50	~85	~5	~10	0	0	0	~2,900	[22]
Versegères	~0.6	15	19.0	18.1	61.0	1.2	0.7	30	2,065	[20]
Calancasca Arvigo	~4	108	34.3	20.9	43.2	1.0	0.6	70	1,965	[20]
Erlenbach	0.027	0.76	0.0	33.7	61.3	5.0	0.0	0	1,348	[23]
Rhine Dipoldsau	244	6,116	20.2	23.0	46.0	8.1	2.8	1,500	1,790	[23]
Rhône Porte-du-Scex	190	5,236	41.6	16.9	32.6	6.0	2.8	1,500	2,123	[23]
Inn S-Chanf	24	617	47.6	6.2	41.3	3.2	1.7	630[a]	2,462	[23]
Ticino Riazzino	71	1,617	19.3	33.8	40.9	3.1	2.9	1,000	1,635	[23]
Thur Andelfingen	48	1,704	1.5	25.6	18.4	45.7	8.8	6,300	768	[23]

[a]Without seasonal guests

Notation for land uses:

Barren: land with no vegetation, e.g. snow fields, glacier, bedrock, rock debris, surface waters

Forest: dense forest, can be exploited, not fertilized

Ext. used: land covered with vegetation, which is not or only scarcely used and not fertilized

Int. used: agricultural land that is fertilized strongly, e.g. arable land, pastures, and vineyards

Settl.: human settlements, e.g. villages, roads, and industrial areas

The six glacier rivers are situated pairwise in three regions, the Val Roseg in Graubünden, Alps of Uri, and the Bernese Alps, their land use is a rough estimate

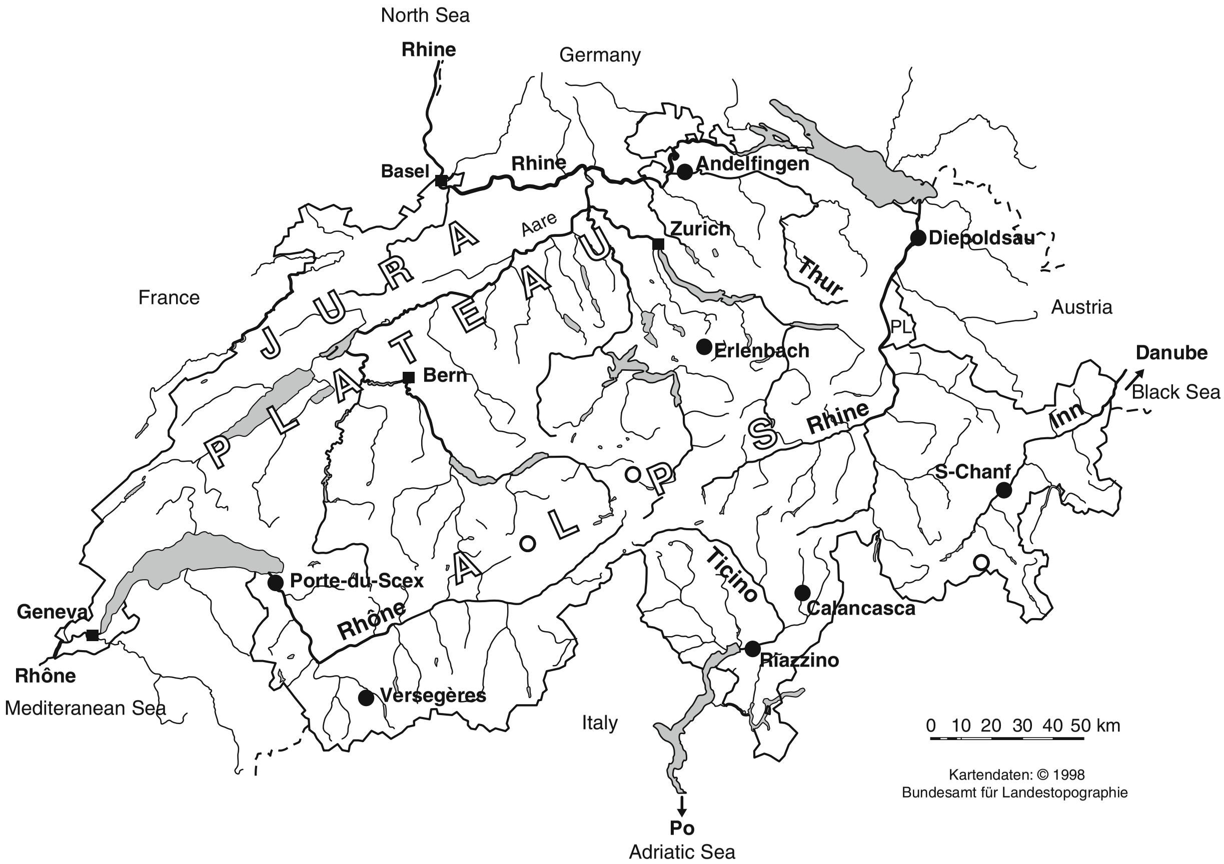

Fig. 3 Map of Switzerland and river stations from which chemical data are used in this contribution; glacier streams, *circle* only. Running waters from the Alps flow in all four directions and in four different seas

1.4 Objectives

The following sections aim to illustrate the key words displayed in Fig. 2 using measured data and to discuss the important processes regulating the observed water conditions. The change in nutrient concentrations in Swiss alpine running waters will be reported proceeding from high alpine regions to lower alpine valleys. Data gained from the long-term monitoring NADUF program facilitate the identification of significant trends in nutrient concentrations and loads as well as geochemical parameters. The reasons for observed trends will be discussed in the framework of water pollution control measures taken, and climate change. Data used also enable us to calculate export coefficients for nutrients characterizing the different land uses and to estimate geochemical weathering rates.

2 Current Chemical Conditions

2.1 Phosphorous

Concentrations of dissolved reactive phosphorous (DRP) (Fig. 4) in glacier streams, and the two small streams Calancasca and Versegères, fell in the range of a few

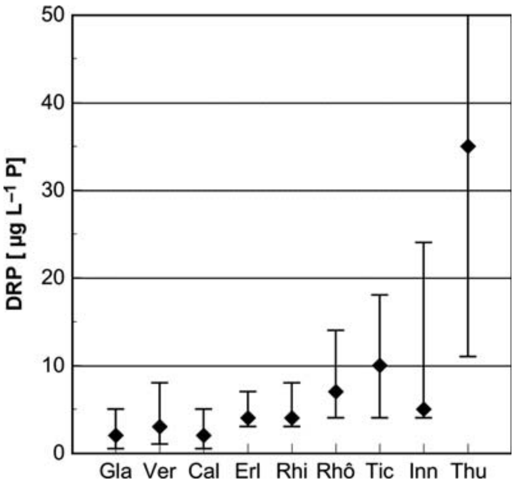

Fig. 4 Concentration ranges (5%, 50%, and 95% percentiles) of DRP in six glacier streams (Gla), two alpine streams Versegères and Calancasca (Ver, Cal), one prealpine stream Erlenbach (Erl), three alpine rivers Rhine (Rhi), Rhône (Rhô), and Ticino (Tic) and the midland river Thur (Thu). Note: Database: glacier streams altogether 90 grab samples (1998/99), Versegères 27 grab samples (1980/81), Calancasca 34 grab samples (1980/81), Erlenbach 52 continuous and flow proportional weekly samples (2005), Ticino continuous and flow proportional biweekly samples 1997–1999 Rhine, Rhône, and Thur continuous and flow proportional biweekly samples 2001–2005

μg L^{-1} P. Values were just above the analytical detection of the method applied in these three surveys. Human activities in these catchments are very small; therefore concentrations measured reflect natural background values of alpine running waters in the Swiss Alps. Concentrations can be considered being due to the weathering of phosphate-containing minerals and small atmospheric inputs. The lower average altitude of the Versegères catchment may explain the somewhat higher concentrations of DRP in this stream in relation to those in the glacier streams and Calancasca stream; that is by more active vegetation. The increase in average DRP in the Erlenbach stream to 4 μg L^{-1} P compared to the three other rivers may also be caused by the higher detection limit of the analytical method applied in this survey. The average DRP concentrations in the large rivers Rhine, Rhône, Ticino, and Inn were still low considering the existing inputs from treated wastewater and runoff from agricultural land. These low average DRP concentrations result from the water protection measures taken in Switzerland to decrease phosphorous (P) inputs into surface waters [24], especially into lakes where eutrophication occurred [25]. Trend analysis of P concentrations in Swiss rivers [26], including alpine rivers, exemplified in Fig. 5, have clearly shown the decrease in P loads. The high content of SS also favors low concentrations of dissolved P in alpine rivers due to sorption of P to particles. Therefore, in rivers exhibiting high concentrations of fine particles, the potential bioavailable load of P can be underestimated by using measured DRP data [27]. Figure 4 also clearly displays the difference in the DRP concentration between natural rivers and slightly impaired running waters and the River Thur, which lies in a catchment on the Swiss Plateau where nearly half of the catchment area is used by intensive agriculture and the input of treated wastewater is significant.

Concentrations of total phosphorous (TP) mostly exceeded those of DRP and exhibit a different pattern (Fig. 5). During high flow conditions, concentrations of TP and SS both show high values that indicate a good correlation between these two chemical parameters. As an extreme, the weekly sample during a high flow event

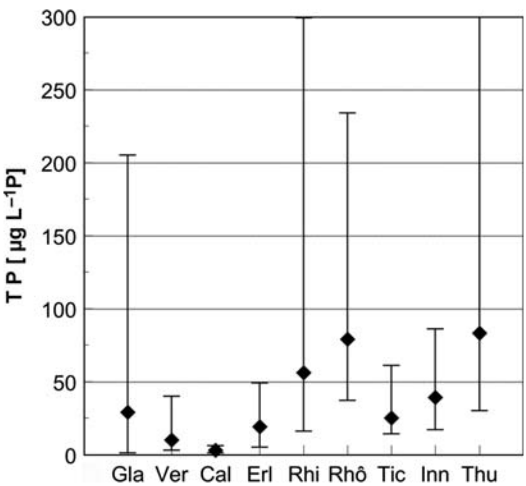

Fig. 5 Concentration ranges (5%, 50%, and 95% percentiles) of total phosphorous (TP) in five glacier streams (Gla), two alpine streams (Ver, Cal), one prealpine stream (Erl), three alpine rivers Rhine (Rhi), Rhône (Rhô), and Ticino (Tic) and the midland river Thur (Thu). Database see legend for Fig. 4

that occurred in the alpine Rhine in August 2005 exhibited a TP concentration of 2,550 μg L^{-1} P and a SS concentration of 2.52 g L^{-1}. This means that 78% of the yearly load of TP and 63% of the SS were discharged during this peak flow. Speciation data from the glacier streams and the Versegères and Calancasca [20, 22] showed that the total dissolved fraction of phosphorous is only slightly higher than the DRP concentration, but clearly less than the TP. Therefore, it can be assumed that the main fraction of the P load present as TP measured in alpine rivers is not bioavailable to algae when these waters are discharged into lakes.

2.2 Nitrogen

Nitrate (NO_3) concentrations (Fig. 6) observed in the glacier streams and the Versegères, Calancasca, and Erlenbach reflect the contemporary background values in alpine running waters. Concentrations are mainly due to atmospheric inputs of inorganic nitrogen compounds such as ammonium (NH_4) and NO_3. The higher atmospheric nitrogen deposition rate in the southern Alps than in the northern region [28] may explain the higher NO_3 concentration in the Calancasca stream. Concentrations in the alpine rivers, Rhine to Inn, exceeded background values, indicating a small but clear anthropogenic impairment. NO_3 concentrations in the Thur were above those measured in alpine running waters. They indicate the high nitrogen inputs from fertilized agricultural lands [29].

The median ammonium concentration in the glacier streams, the Versegères and the Calancasca equaled 10–15 μg L^{-1} N, whereas ammonium levels mostly went over the median in summer. The median values of nitrite (NO_2) were low in the range of 1–2 μg L^{-1} N. Concentrations of dissolved organic nitrogen in glacier streams were about one third of that of NO_3. Therefore, ammonium and nitrite contributed little to the background concentration of total dissolved nitrogen

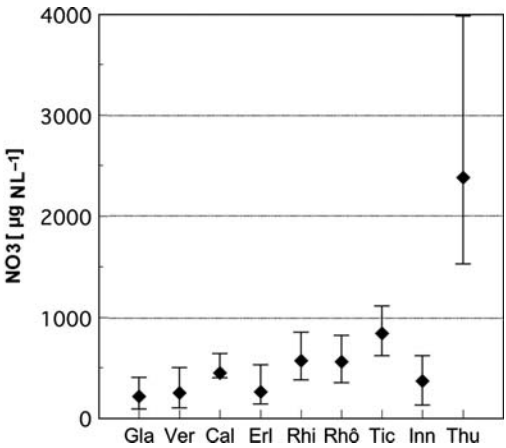

Fig. 6 Concentration ranges (5%, 50%, and 95% percentiles) of nitrate in six glacier streams (Gla), two alpine streams (Ver, Cal), one prealpine stream (Erl), three alpine rivers Rhine (Rhi), Rhône (Rhô), and Ticino (Tic) and the midland river Thur (Thu)

content in natural alpine rivers. In addition, concentrations of nitrite and ammonia, that is produced in the pH-dependent equilibrium with ammonium, remained well below the toxicity level for salmonoids in fresh waters [30]. The content of particulate nitrogen in glacier streams was <10% of that of NO_3 or total nitrogen. In the alpine rivers, Rhine to Inn, NO_3 amounted to 70–85% of the total nitrogen [26]. In the case of the Thur, nitrate made up over 90% of the total nitrogen. This means that the other nitrogen fractions mentioned above contribute little to the total nitrogen load in Swiss rivers.

2.3 Other Nutrients

In waters, silicic acid (H_4SiO_4) represents an important nutrient for diatoms. Potassium (K) also acts as a nutrient for aquatic plants, although it is rarely limiting. Their concentrations in alpine rivers are strongly governed by the chemical weathering of silicates (rock-forming minerals) such as biotite, K-feldspar, and clay minerals. Concentrations of K and H_4SiO_4 observed in alpine rivers (Fig. 7) reflect qualitatively the relative abundance of silicates in the catchment. Highest values, medians and 95% percentiles, were reported in the Ticino, a catchment covered mainly with silicates, gneiss, and granites. The lowest concentrations were observed in the Erlenbach, a catchment exhibiting calcareous rocks, mainly flysch (calcareous sandstones with clay-rich schists), which also contain some clay. The relatively high range of K in the Thur, a catchment dominated by calcareous rocks, mainly molasses, can be explained by inputs from fertilized agricultural lands.

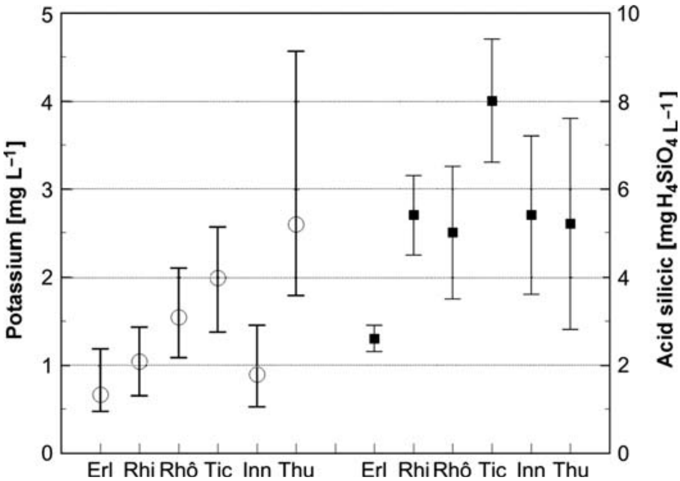

Fig. 7 Concentration ranges (5%, 50%, and 95% percentiles) of potassium and silicic acid in one prealpine stream (Erl), three alpine rivers Rhine (Rhi), Rhône (Rhô), and Ticino (Tic) and the midland river Thur (Thu)

3 Trends

3.1 Introduction

The long-standing data series obtained by the NADUF program allowed the evaluation of changes and trends in nutrient concentrations and loads. The mathematical model chosen describes the time series in concentration and load as a yearly sinusoidal function, representing the seasonal fluctuation and the linear change with time. The regression calculation with the data measured gives the average yearly trend and the average amplitude in seasonal variation. In addition, the calculation also estimates the standard deviations of the regression coefficients [26]. Trends and amplitude are considered significant if the correlation coefficient of the regression is statistically significant at the 95% level and the 95% confidence intervals of the linear trend or amplitude do not include zero. This kind of time-series analysis allows detecting small but significant linear trends down to 0.1% per year depending on the chemical parameter and water flow regime.

3.2 Phosphorous

The long-term time series in DRP concentrations, exemplified in Fig. 8, showed a clear decrease in most Swiss rivers and a downward step in 1986, the year in which

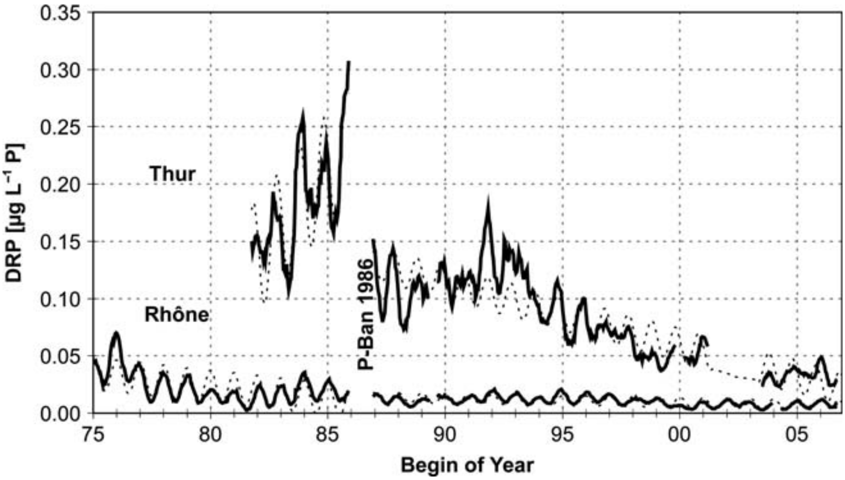

Fig. 8 Influence of the P-ban in detergent to the moving averages over a period of 22 weeks (*solid line*) for DRP concentration in the Thur and Rhône. Sinusoidal regression curves (*dashed line*) before and after the ban

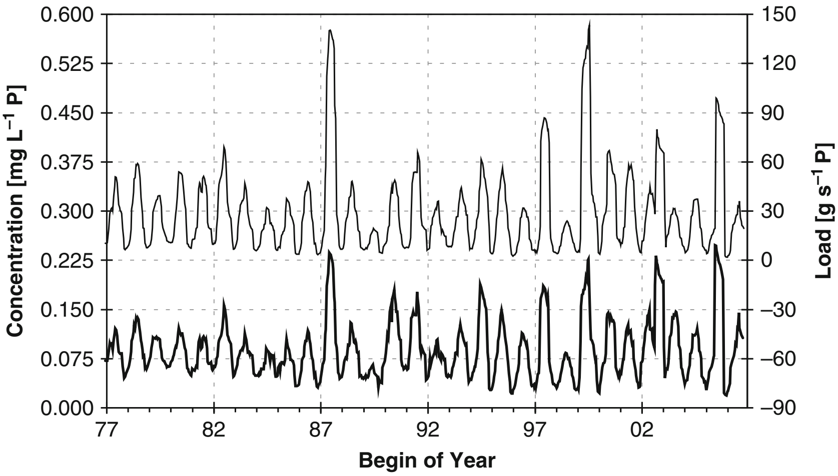

Fig. 9 Moving average over a period of 22 weeks for TP concentration (*lower curve*) and load (*upper curve*) in the Rhine from 1977 to 2006

the P-ban in detergent was enforced. The yearly decrease in the Rhône amounted to 2.3 μg L^{-1} P in the period 1975–1985, it decreased to 0.3 μg L^{-1} P afterwards. In the River Thur, an increase was noted in the short measuring period before the P-ban. After 1986, the DRP concentration clearly decreased at a yearly rate of 5.5 μg L^{-1} P. However, in the last two years the downward trend appeared to level out. In alpine rivers, decreases in DRP concentrations and loads were usually small but mostly statistical significant [26]. Trends estimated in the alpine Rhine are similar to those in the Rhône.

TP concentrations and loads displayed strong seasonal fluctuations (Fig. 9). In addition, peak flow resulted in high TP concentrations and loads. The trend analysis yielded no statistically significant change over the measuring period of 30 years in the Rhine or 32 years in the Rhône. However, at shorter intervals, some significant changes could be recognized, such as in the period from 1977 to 1986.

Siegrist and Boller [24] estimated that the P-ban in detergents resulted in a decrease of P input of 2,600 t P y^{-1} (390 g P s^{-1}) by wastewaters into Swiss surface waters. Referring this decrease to the alpine Rhine catchment (proportional to the number of inhabitants), a reduction in P-load of 145 t y^{-1} (4.5 g s^{-1}) can be expected, giving an average concentration decrease of 18 μg L^{-1} P. For the alpine Rhône, this decrease would equal 110 t y^{-1} P (3.6 g s^{-1} P), resp. 19 μg L^{-1} P and for the Thur 120 t y^{-1} P (3.7 g s^{-1} P), resp. 77 μg L^{-1} P. The measured decreases in P in the alpine Rhine and Rhône (Fig. 8) were not as large as those expected by the P-ban, although the P input by detergents in wastewaters occurs mainly in the form of DRP. The higher decrease found in the Thur may be attributed to P control measures taken on agricultural lands and in enhanced P elimination in wastewater treatment plants. Two factors may explain why the effect of the P-ban could not be clearly detected by monitoring TP: The decrease in the P load in the alpine rivers due to the P-ban was small compared to the total measured TP load, and the strong yearly fluctuation in measured TP loads and concentrations may mask the decrease. The main source of TP in alpine running waters originates from soil particles that are flushed during heavy rainfalls or by snowmelt into running waters.

3.3 Nitrate

NO_3 concentrations and loads in the Rhône, as well as in the Rhine, showed clear trends (Fig. 10). From the mid-1970s to the end of the 1980s, both concentrations and loads increased significantly in the range of 2–5%, that is 9–14 μg L^{-1} N per year. From the 1990s to 2006, concentrations and loads decreased significantly in the Rhône, about 0.5% per year, whereas decreases in the Rhine were insignificant. This downward trend of NO_3 is probably due to a change in agricultural practices, as they adapted the input of fertilizer to the need of the crops and avoiding fallow fields in winter. During the entire measuring period from 1975 to 2006, NO_3 concentrations increased significantly, 0.4% per year in the Rhône and 0.2% per year in the Rhine. The NO_3 loads did not change significantly in either river, although the water discharge in the Rhine decreased 0.5% per year.

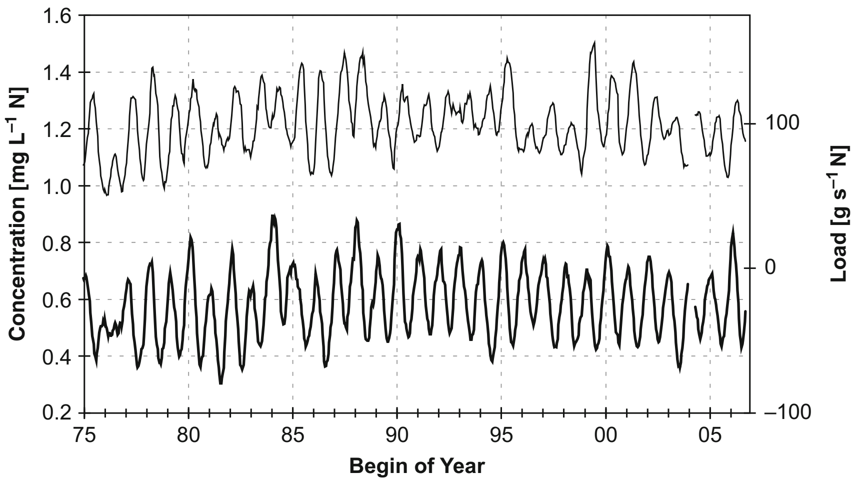

Fig. 10 Moving average (period 22 weeks) of nitrate concentration (*lower curve*) and load (*upper curve*) in the Rhône from 1975 to 2006

3.4 Water Temperature and Geochemical Parameters

Water temperatures measured continuously at the sampling stations of the Rhine and Rhône have increased significantly. In the River Rhine, the increase amounted to 0.031°C or 0.43% per year or 0.95°C in 30 years (Fig. 11), and in the River Rhône the average increase was 0.023°C or 0.34% per year or 0.69°C in 30 years. In rivers on the Swiss Plateau [26, 31], temperature increases mostly were over 0.4% per year. Figure 11 displays a remarkable temperature increase in the period 1987–1992 when the water discharge was low. Therefore, one might argue that the temperature increase was due to the change in water discharge. Indeed, the heat flux in the Rhine as well as in the Rhône did not change significantly over the measuring period from 1975 to 2006. However, this result would be inconsistent with the warming effect of running waters expected by increased air temperature (see [2], this volume). When discussing the causes of water temperature changes in alpine rivers, one must consider that temporal and spatial changes in water flow provoked by hydroelectric power stations could also influence the water temperature regime to an extent not yet evaluated [31].

In the context of the discussion raised in the literature on the impact of global warming on weathering [e.g. 12, 15, 18], it will be interesting to evaluate the concentration and load changes of geochemical constituents in alpine rivers. Time series analysis described in Sect. 3.1 allows the analysis of their changes in the Rhine and Rhône River. Measuring periods in the other alpine rivers cited were too short to yield meaningful results.

Quantitatively, the most important weathering product, bicarbonate expressed as alkalinity, exhibited a small, but statistically significant, decrease of 2.6 μmol L^{-1}

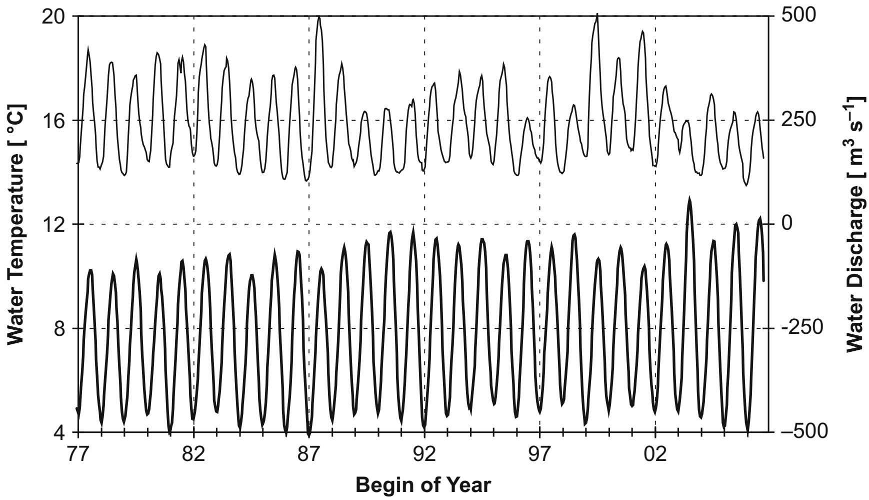

Fig. 11 Moving average (period 22 weeks) of water temperature (*lower curve*) and water discharge (*upper curve*) in the Rhine

per year; equivalent to 0.13% of the alkalinity observed in the Rhine and 0.20% of that in the Rhône. These declines can be explained quantitatively by the diminishing solubility of calcite with increasing temperature [32]. The load of alkalinity in the Rhine diminished slightly by 0.32% per year, caused by the decrease in the water discharge of 0.47% per year from 1977 to 2006. In contrast, the alkalinity load in the Rhône exhibited no change. The main ionic counterpart of bicarbonate in natural waters, calcium and magnesium, exhibited a different behavior. Mg concentrations increased notably and significantly; 1.5 μmol L^{-1} per year or 0.72% in the Rhône and 2.3 μmol L^{-1} or 0.72% in the Rhine. Mg loads also became greater in both rivers, as has been observed in most rivers monitored in the NADUF programme [26]. It can be hypothesized that the observed increase is due to the release of Mg from soils brought in by applied fertilizers containing Mg as a minor ingredient. Ca concentrations, on the other hand, decreased insignificantly by 0.34 μmol L^{-1} per year in the Rhone and significantly by 1.1 μmol L^{-1} per year in the Rhine. In general, these decreases were consistent with the temperature dependence in calcite solubility. Calcium loads decreased significantly 0.19% per year in the Rhône and 0.52% in the Rhine. The opposite time trend of the two earth-alkaline ions observed in the Rhône yielded no significant patterns in concentration and load of the sum of both ions, that is total hardness. With regard to the Rhine River, the concentration in total hardness increased slightly and the load decreased as caused by the decline in water discharge.

The opposite trend in concentrations of the earth-alkaline ions induced a remarkable and statistically significant change in the magnesium to calcium ratio (Fig. 12). It is worth mentioning that the seasonal minimum in the ratio occurs in summer when Mg and Ca also show their minima. The relatively larger seasonal decrease in

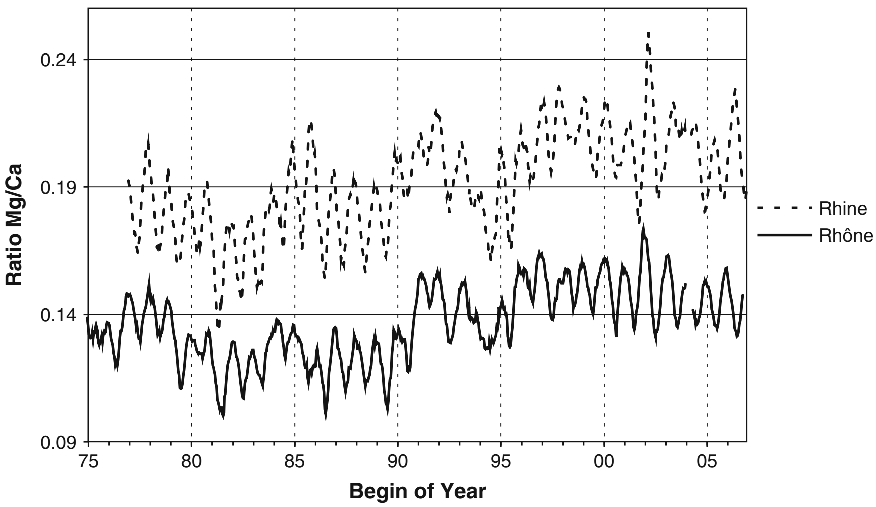

Fig. 12 Moving average over a period of 22 weeks of the molar ratio Mg to Ca in the Rhône and Rhine from the 1970s to 2006

Mg than in the Ca concentration causes the summer minimum in the ratio. Mg to Ca ratios did not rise evenly over the measuring period. In the years 1987–1997, a clear increase occurred. In the last 7 years, changes were leveling. The increase in the Mg to Ca ratio has been observed in most of the other rivers monitored by the NADUF program [26].

From 1975 to 2006, the concentration of H_4SiO_4, the weathering product of silicates and clays, diminished slightly but significantly by 0.13 μmol L^{-1} or 0.23% per year in the Rhône and nonsignificantly in the Rhine (0.03 μmol L^{-1}). Loads of H_4SiO_4 decreased significantly by 0.6% per year in both rivers. The calculated downward trends in H_4SiO_4 are contradictory to expected increases in the weathering rate of silicates due to observed increases in air and water temperatures.

K, a weathering product and fertilizer ingredient, exhibited a strong increase in all Swiss rivers. In the Rhône, this amounted to 17 μg L^{-1} or 1.5% per year and 9.4 μg L^{-1} or 1.1% per year in the Rhine. Trends in loads were also highly significant. This large increase in K can be best explained by a small release from the large K deposit built up in soils fertilized over the last few decades [26].

4 Mass Fluxes

4.1 Export Coefficients

Export coefficients per unit area of the catchment and per unit time characterize the mass flow (load) of a river water constituent at the outlet of the catchment. For conservative constituents, that is for which no relevant biogeochemical transformation processes are occurring in the river water, the export at the outlet is equal to the sum of all inputs into the river coming from the various point and diffuse sources. This assumption can be regarded as a valuable approximation in alpine rivers for most of the chemical parameters discussed. Export coefficients allow comparing different catchments in size, land use, or other characters describing a basin.

DRP, NO_3, and TP export coefficients listed in Table 3 exhibited about the same relative size pattern between rivers like their concentrations depicted in Figs. 4, 6, and 7, since specific water discharges between rivers fell into the same size range. DRP and NO_3 coefficients calculated for the small and remote catchments (glacier streams, Versegères, Calancasca, and Erlenbach) did not vary significantly. These differences between the four catchments cannot be related to the differences in land cover reported in Table 2. A Bayesian estimation [29] of DRP and NO_3 export coefficients for each kind of the five input sources stated in Table 2, using the whole long-term NADUF data set, gave very similar values for the three categories of land cover (barren land, dense forest, and extensively used land), but with large confidence intervals. In these calculations, mean values for DRP varied between 4 and 6 mg P m^{-2} y^{-1} and those of NO_3 between 360 and 480 mg N m^{-2} y^{-1}, when taking the load calculation mode. Coefficients for intensively used agricultural land

Table 3 Export coefficients for water, nutrients, and suspended solids

Parameter	Water discharge	NO_3	DRP	TP	TP	SS	SS
Unit	$m^3\ m^{-2}\ y^{-1}$	$mg\ m^{-2}\ y^{-1}$	$mg\ m^{-2}\ y^{-1}$	$mg\ m^{-2}\ y^{-1}$	$mg\ m^{-2}\ y^{-1}$	$g\ m^{-2}\ y^{-1}$	$g\ m^{-2}\ y^{-1}$
River	Mean	Mean	Mean	Mean	Median	Mean	Median
Gla	~1.5	~350	~3	~110	n.d	~125	n.d
Ver	1.26	280	4.4	20	n.d	n.d	n.d
Cal	1.17	540	2.5	3.4	n.d	n.d	n.d
Erl	1.31	400	4.2	65	31	106	27
Rhi	1.17	619	3.8	167	55	229	51
Rhô	1.24	663	7.8	166	88	180	60
Tic	1.53	1,210	14.2	47	25	25	7
Inn	1.08	312	5.2	63	12	36	1
Thu	0.86	2,240	36.5	141	57	73	14

n.d.: no data, river abbreviation see legend Fig. 4

Database: For glacier streams, Versegères and Calancasca: mean load calculated as mean discharge times mean concentration. For the other rivers, loads represent means and medians, respectively, of biweekly water discharge proportional samples over the whole measuring period were distinctly higher, 41 mg P m^{-2} y^{-1} and 2,900 mg N m^{-2} y^{-1}, respectively. The contribution of inhabitants, including settlements, amounted to 0.16 kg P $inhab^{-1}$ y^{-1} and 2.1 kg N $inhab^{-1}$ y^{-1}, giving 9.6 mg P m^{-2} y^{-1} and 126 mg N m^{-2} y^{-1} for the most populated alpine catchment, the alpine Rhine. The sum of the five Bayesian partial coefficients fitted reasonably well with the measured NO_3 export coefficient for the larger and populated alpine catchments listed in Table 3. In contrast, most observed DRP coefficients fell below the sum of the Bayesian coefficients. This effect might be due to the partial sorption of DRP on SS. Observed DRP and NO_3 export coefficients for the Thur catchment, with its high percentage of agricultural land and higher population density, clearly exceeded those of the other catchments.

TP export coefficients showed a different pattern. They were strongly related to those of SS, that is turbidity. This flow is governed by the physiographical condition of the catchment and the type of bedrock. Mean and median values of TP and SS export coefficients differed greatly. This effect is due to the very high concentrations measured during periods of high water discharge and peak flows.

4.2 Weathering Rates

Weathering rates of rock-forming minerals can be estimated by using a simple input–output box model, whereas the atmospheric deposition [26] represents the input into the catchment and the output are the loads measured at the outlet of the basin, that is at the measuring station. The difference between input and output is assigned to the products formed by chemical weathering taking place with rock-forming minerals in soils and aquifers, represented by the dissolved water constituents. Conversion processes of geochemical parameters occurring in rivers are neglected since they are considered small. Rates of minerals weathered are calculated by the aid of the stoichiometric relations given in Table 1. The exact reconstruction procedure is described in [10] and applied in [26]. Average loads over the measuring period at each station were deployed to estimate weathering rates depicted in Fig. 13.

Weathering of sedimentary rocks, containing calcite, dolomite, and anhydrite, clearly dominated in all catchments (Fig. 13), and also in the Ticino and Inn catchment where bedrock of silicates, mainly granites and gneiss, cover a major part of the basin. This effect is due to a much lower dissolution rate per unit area of land surfaces or mineral surfaces per unit time of silicates compared to calcareous rocks [19]. Meybeck [33] found that the world average relative chemical erosion rate of carbonate rock is 12 times and that of gypsum or anhydrite is 40 times higher than the weathering rate of granite or gneiss.

Large differences in the proportion of silicate bedrock in catchments presented only weakly influenced the silicate-weathering rate for the whole basin (see silicates bar in Fig. 13). In the Ticino basin, mostly covered with silicate bedrock, the rate increased to 100 mmol m^{-2} y^{-1} (40 g m^{-2} y^{-1}). This value was in the range

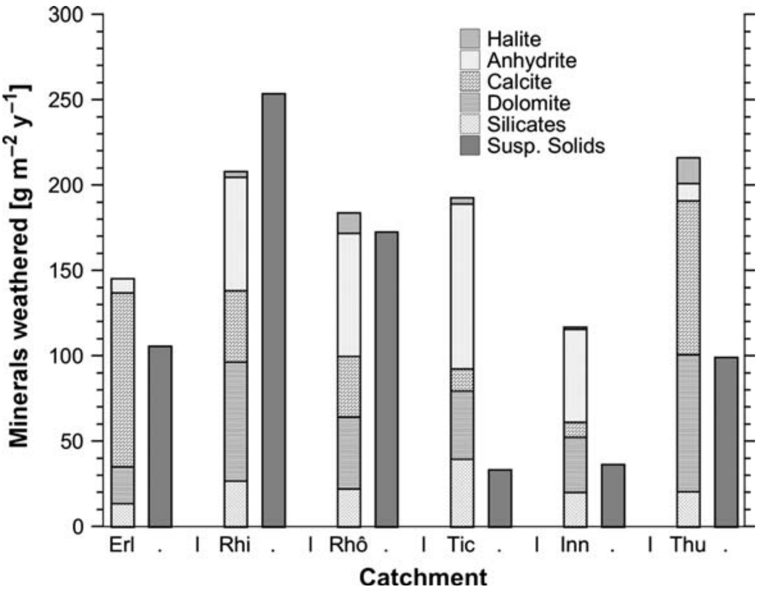

Fig. 13 Minerals weathered chemically and suspended solids weathered physically in alpine catchments and Thur basin. Abbreviation of rivers, see legend for Fig. 4

found in glacier streams in alpine crystalline catchments [34]. In the Rhine and Rhône catchments, silicate-weathering rates were 68 and 56 mmol m^{-2} y^{-1} (27 and 22 g m^{-2} y^{-1}). In the Erlenbach and Thur catchments, having no silicate bedrock, silicate-weathering rates estimated from the load of silicic acid, H_4SiO_4, amounted to 35 and 52 mmol m^{-2} y^{-1}, respectively. This inconsistency can be resolved by assuming that carbonate lithologies also contain clays and quartz in the rock and overlying soils, which will be weathered. Indeed, Bluth and Kump [16] have shown that a carbonate lithology also exhibits a clear flux of H_4SiO_4. Taking their empirical relationship between the flux of H_4SiO_4 and water runoff gained from 101 river basins, each draining a primary lithology, the Thur, as a basin with carbonate rocks, would have a silicate chemical denudation rate of 113 mmol m^{-2} y^{-1} and the Ticino, taken as a granitic basin, would exhibit a rate of 194 mmol m^{-2} y^{-1}. These rates estimated by the relationships shown in [16] clearly exceed those measured in rivers displayed in Fig. 13. The positive relationship between chemical weathering rates and runoff found by Bluth and Kump [16] also highlights the importance of the formation of new reactive particles by physical weathering as regulating factors for chemical weathering rates.

In contrast to the fate of silicates, a catchment exhibiting a small area of so-called "Bündner Schiefer," a sandy-marly schist containing soluble anhydrite or gypsum, will produce a remarkably high weathering rate for the entire basin. This effect arises in the alpine catchments of the Ticino, Rhine, and Rhône. The occurrence of "Bündner Schiefer" also causes sulfate concentrations in the range of 0.5–1 mmol L^{-1} in rivers. Natural and anthropogenic atmospheric sulfur deposition [28], as well as pyrite oxidation, contribute little to the sulfate concentration in Swiss rivers [26]. Weathering rates calculated for halite indicate primarily the anthropogenic input of sodium chloride from its diverse uses such as road salting, use in ion exchangers in water-softening equipment, and in animal and human wastes. In the Erlenbach catchment, exhibiting negligible pollution sources, the weathering rate of halite fell to zero. Observed chloride concentrations in the Erlenbach stream were in the range of 0.5 mg L^{-1}; denoting atmospheric wet and dry deposition.

Total chemical weathering rates reported in Fig. 13 varied between 119 and 212 g m^{-2} y^{-1}. They correspond to annual denudation rates between 0.045 and 0.075 mm per year. Rates in the alpine catchments of Rhine, Rhône, and Ticino clearly exceeded those estimated for the whole basin (River Po in Ticino) by Gaillardet et al. [18]; they cite 53, 109, 126 g m^{-2} y^{-1}, respectively. Chemical weathering rates of alpine rivers were well above the world average, 24 g m^{-2} y^{-1} cited in [18] or 23 g m^{-2} y^{-1} given in [13]. This fact highlights the importance of runoff and the formation of new reactive particles by physical weathering as regulating factors for chemical weathering rates. Alpine rivers discussed here exhibited slightly higher chemical weathering rates than alpine rivers in the Himalaya [34, 35].

In the catchments of the large alpine river Rhine and Rhône and the Erlenbach stream, chemical and physical weathering rates estimated from SS loads fell in the same range. Physical weathering rates in the Inn and Ticino basin were even smaller than the corresponding chemical rates. Physical weathering rates shown in Fig. 13 were below those reported for rivers in the Himalaya [34, 35]. The estimated river inputs of sediments into oceans, the world average physical erosion rate, amounts to 226 g m^{-2} y^{-1} according to [13].

The physical erosion rates shown in Fig. 13 were estimated from data of SS loads. To be correct, they do not represent the total sediment yield transported with the river water. The latter is commonly interpreted as physical erosion or denudation rate. The continuous and water discharge proportional sampling device discriminates larger particles, which contribute strongly to the total load during peak water flows. Therefore, loads of solids gained by this sampling device undervalue up to a factor of three in high flood years the mean sediment transport rate of alpine basins cited in [3] obtained with grab samples. However, estimations of suspended sediment loads based on grab samples also exhibit a distinct uncertainty due to the extrapolation procedure related to periods of peak water flows not sampled [36].

5 Conclusion

The chemical data presented clearly show the special character of alpine running waters compared to those in the midland and lowlands. The high water flow in these systems and the low anthropogenic activities produce only a small impact from pollution of nutrients and results mostly in their good chemical status. This prerequisite for having a good ecological status is hampered by short-term changes in their water flow regime provoked by hydroelectric power production. Nutrient export coefficients in alpine streams fall in the range of those estimated for nonagricultural lands and forests on the Swiss Plateau. Trend analysis for nutrients confirmed the success of permanent efforts to decelerate the anthropogenic cycle of nitrogen and phosphorous in alpine regions. Long-term observations also revealed small changes in concentrations and loads of geochemical constituents in alpine rivers and therefore slightly affecting chemical weathering rates. The measured gradual temperature increase is likely one cause for these changes in the geochemical condition. However, other factors also will play an important role for explaining observed changes in concentrations and loads of geochemical constituents.

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Effect of Acid Deposition on Chemistry and Biology of High-Altitude Alpine Lakes

S.M. Steingruber and L. Colombo

Abstract Trend analyses of the key parameters involved in acidification processes measured in 20 Alpine lakes during the period 1980–2004 revealed significant decreasing sulphate (15 out of 20) and increasing alkalinity trends (14 out of 20) in most studied lakes, while trends for base cations and nitrate were small and mostly insignificant. The average increase in alkalinity between 1980 and 2004 was 0.012 meq l^{-1}. Today two lakes out of 20 are still acidic (alkalinity < 0 meq l^{-1}), 13 are sensitive to acidification (0 meq l^{-1} < alkalinity < 0.05 meq l^{-1}) and five have low alkalinities but are not at risk (0.05 meq l^{-1} < alkalinity < 0.2 meq l^{-1}). Differently, in the 1980s four lakes were acidic, 14 were sensitive to acidification and two had low alkalinities. During the same time period the pH increased on average by 0.3 units.

Because of accelerated dissolution of aluminium minerals a pH value below 6 can become critical for the biology of lakes. Compared to seven lakes in the 1980s today only three lakes out of 20 exhibit an average pH below 6. A comparison between the populations of macroinvertebrates in lakes with different acidity showed that at average lake pH's below 6 the population of macroinvertebrate changes. The taxa richness and the EPT index (= number of families of the order *Ephemeroptera*, *Plecoptera*, *Trichoptera*) decreases and acid-sensitive species disappear.

Keywords Alpine lakes, Acidification, Acid deposition, Long-term trends, Recovery

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Contents

1 Introduction

Atmospheric deposition at the southern side of the Central Alps is mainly determined by warm, humid air masses originating from the Mediterranean Sea and colliding with the Alps. While passing over the Po Plain these air masses become enriched with anthropogenic air pollutants like sulphur dioxide, nitrogen oxides and ammonia. As a consequence atmospheric deposition becomes acidic, which is particularly critical in areas whose geology is dominated by base-poor rocks with low buffering capacity, because it can cause acidification of freshwaters and soils. Acidification of high-altitude Alpine lakes situated in granitic areas was first reported in the early 1980s [1].

1.1 Origin and Effects of Acid Deposition

"Acid rain" is a broad term used to describe several ways that acids fall out of the atmosphere. A more precise term is acid deposition, which has two components: wet and dry. Wet deposition refers to rain, fog and snow, while dry deposition refers to gases and aerosol particles. The primary causes of acid deposition are emissions of sulphur dioxide and nitrogen oxides (SO_2 and NO_x) from combustion of fossil fuels. In the atmosphere these gases can oxidize to sulphate and nitrate acids causing acid precipitations. Besides the above-described mechanism, due to intensification of agriculture, also emissions of ammonia (NH_3) contributes to acid precipitation. Although ammonia itself reacts as a base in the atmosphere (NH_4^+), during assimilation by plants the temporarily bound proton is released to the environment again. In addition, in soils and waters ammonia can be oxidized by microorganisms to nitrate, liberating two protons. In this way, ammonia can also contribute to the acidification of soils and waters. Acid deposition first begun with the industrial revolution, when large amounts of fossil fuels were burnt to produce steam power necessary to drive machinery. The term “acid rain” was coined in the nineteenth century by the scientist Robert Angus Smith, working at the time in Manchester [2]. In those times acid rain was confined to industrial towns and cities. However, the situation gradually worsened and widespread environmental damage on a global scale was observed by scientists in the second half of the twentieth century. In the 1960s the relationship between sulphur emissions in continental Europe and acidification of Scandinavian lakes was demonstrated [3] and between 1972 and 1977 several studies confirmed the hypothesis that air pollutants could travel several thousands of kilometres before deposition and damage occurred, evidencing that cooperation on an international level was necessary to solve problems such as acidification [4]. Since 1979 50 Governments, Switzerland included, signed the Convention on Long-range Transboundary Air Pollution. The Convention entered into force in 1983 and it has been extended by eight specific protocols. The Helsinki Protocol of 1985 aimed at the reduction of sulphur emissions by at least 30%, freezing of the emissions of NO_x was the goal of the Sofia Protocol in 1988, the Oslo Protocol of 1994 required further reduction of sulphur emissions and the Gothenburg Protocol of 1999 set the ceilings for 2010 for the emissions of sulphur, NO_x, VOC’s and ammonia. As a consequence, in the last 20–25 years there has been a substantial reduction in the emissions of sulphur and in some countries also of nitrogen oxides [5] improving the quality of atmospheric deposition. In fact, in the last 20 years a decrease in sulphate deposition and therefore also in deposition of acidity has been observed in many European sites [6].

Figure 1 shows the emissions of sulphur dioxide, nitrogen oxides and ammonia in Switzerland from 1900 to 2010. The sulphur dioxide and nitrogen oxides emission started to increase steeply after the Second World War. Sulphur dioxide reached its maximum between 1965 and 1980, while nitrogen oxides peaked around 1985. Afterwards, both sulphur and nitrogen oxides decreased continuously until 2000. For ammonia almost no decrease can be observed. The reduction of sulphur dioxide emissions has mainly been caused by a reduction of the sulphur content in heating oils and the partial substitution of sulphur-rich carbon with other fossil fuels. The inversion of the tendency of the nitrogen oxide emissions after 1985 was mainly determined by the application of catalytic converters to cars. However, because of its particular meteorology the air quality in Southern Switzerland is not only influenced by local emissions but also by transboundary air pollution originating from the Po Plain and particularly from the urban area of Milan. In fact, wet deposition in Southern Switzerland is mainly determined by warm, humid air masses originating from the Mediterranean Sea, passing over the Po Plain and colliding with the Alps. Similarly to SO_2 emissions, atmospheric SO_2 concentrations [8] and average sulphate rain water concentrations and deposition in Southern Switzerland decreased in the last 20 years. Differently, reduction in nitrogen can only be observed for emissions and local atmospheric concentrations of NO_x [8]. In fact, concentrations of nitrate in rain water and its deposition did still not decrease. The same can be observed for ammonia (Fig. 2).

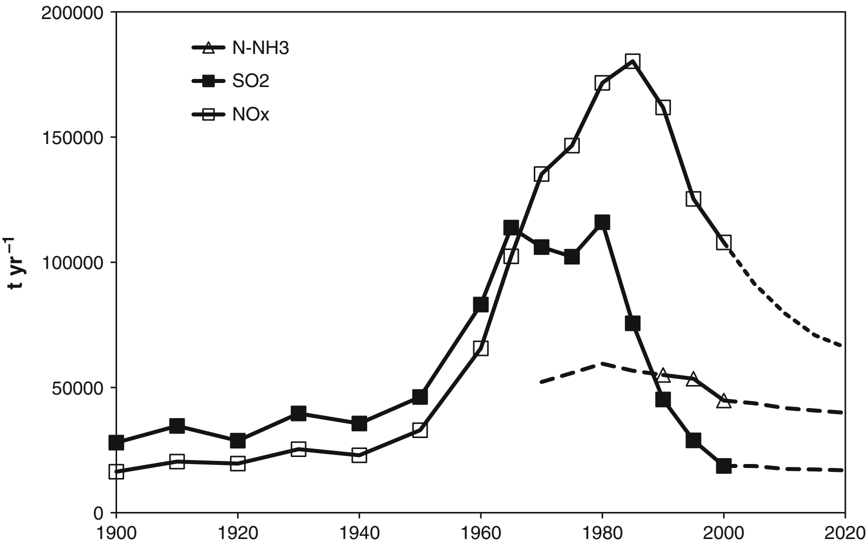

Fig. 1 Annual sulphur dioxide, nitrogen oxide and ammonia emissions in Switzerland from 1900 to 2010. *Dashed lines* indicate estimate values
Source: [7]

1.2 Mechanisms and Effects of Freshwater Acidification

Acid deposition can cause acidification of soils and waters. Acidification can be defined as a reduction of the acid-neutralizing capacity of soils or waters. The acid-neutralizing capacity is also known as alkalinity (Alk) and indicates the sensitivity of a water system toward acidification. It can be expressed as [9]:

$$[\mathrm{Alk}] = [\mathrm{HCO_3^-}] + 2 \times [\mathrm{CO_3^{2-}}] + [\mathrm{OH^-}] - [\mathrm{H^+}] \tag{1}$$

or

$$[\mathrm{Alk}] = [\mathrm{Na^+}] + [\mathrm{K^+}] + 2 \times [\mathrm{Mg^{2+}}] + 2 \times [\mathrm{Ca^{2+}}] + [\mathrm{NH_4^+}] - [\mathrm{Cl^-}] - 2 \times [\mathrm{SO_4^{2-}}] - [\mathrm{NO_3^-}] \tag{2}$$

for pH $<$ 8.2:

$$[\mathrm{Alk}] \cong [\mathrm{HCO_3^-}] - [\mathrm{H^+}] \tag{3}$$

Distilled water, which equilibrates with CO_2 of a non-polluted atmosphere, has a pH of 5.65. At this pH, proton and bicarbonate concentrations are equal and alkalinity becomes zero [9]. We consider freshwaters acidic when [Alk]$<$0 meq l^{-1},

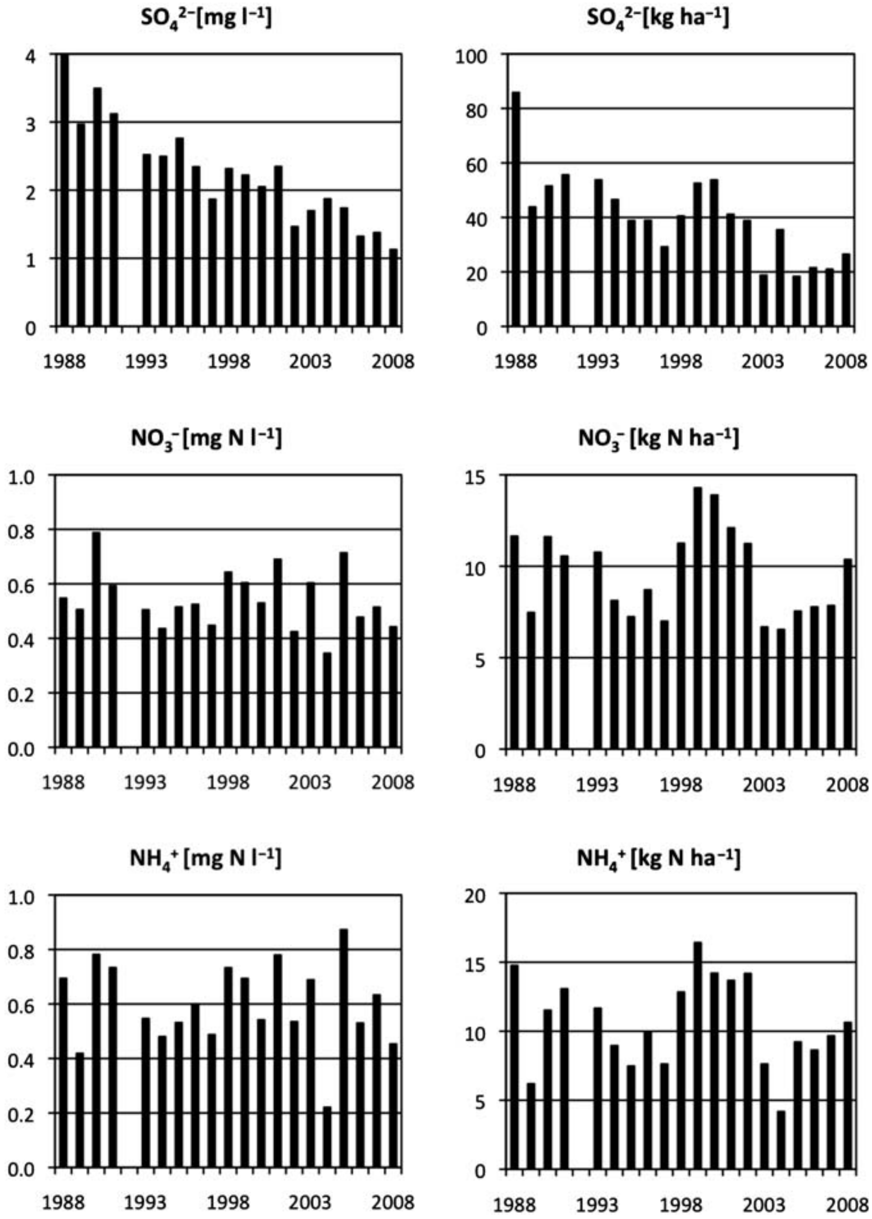

Fig. 2 Yearly average rain water concentrations and wet deposition of sulphate, nitrate and ammonia at Locarno Monti

sensitive to acidification when $0<[Alk]<0.05$ meq l^{-1} and with low alkalinity but not sensitive to acidification when $0.05 <[Alk]<0.2$ meq l^{-1} [10]. The critical alkalinity value, below which acid deposition exceeds the critical load (i.e. the load which will leave a lake alkalinity that ensures fish development), [11, 12] was fixed at 0.02 meq l^{-1}.

Alkalinity is the result of complex interactions between wet and dry deposition and the soil and rocks of the watershed and biologic processes. The involved chemical and biological reactions can be divided into weathering reactions, ion-exchange reactions, redox processes and the formation and degradation of biomass and humus [13]. Weathering reactions consume protons and release base cations (Ca^{2+}, Mg^{2+}, Na^{+}, K^{+}). Carbonates (lime, dolomite) are weathered faster than aluminium silicates or aluminium oxides. Ion-exchange reactions at the surface of clay minerals or humus have the same effect as weathering but are faster (minutes compared to hours or days) and have a smaller pool of exchangeable bases. In the long run, chemical weathering is the rate-limiting step in the supply of base cations for export from the watershed. Reduction processes also increase alkalinity (e.g. denitrification, sulphate reduction), while oxidation reactions (e.g. nitrification, oxidation of hydrosulphide) reduce alkalinity. Production of biomass can increase alkalinity if nitrate is assimilated, while it can decrease alkalinity if ammonia is assimilated. It can be concluded that oligotrophic freshwaters that are poor in carbonates and electrolytes are particularly sensitive to acidification. Such waters can be found in areas with weathering-resistant bedrock (granite, gneiss) with a thin soil surface like some high-altitude Alpine regions in Southern Switzerland.

An important consequence of acidification is mobilization of metals from terrestrial watersheds [14]. Particularly important is the release of aluminium because of its toxic effects on freshwater biota especially on fish [15]. Not all Al forms are toxic. Only cationic species contained within the operational forms termed labile Al (LAl) or inorganic monomeric Al (Ali) are gill-reactive and hence affect fish health [16]. It has been shown that concentrations of soluble aluminium increase with decreasing pH from a pH of ca. 6.3 [17].

Acidification of acid-sensitive waters is accompanied by severe changes in biological communities. Effects range from reductions in diversity without changes in total biomass to elimination of all organisms. In many cases the immediate cause of the changes is unknown. Some effects are the result of H^{+} toxicity itself or of the toxicity of metals mobilized from the watershed, others have more indirect causes such as changes in predator–prey interactions or in physical conditions of lakes (ex. transparency). [14]

Dillon et al. [14] resumed the biological consequences of acidification as follows.

> Decomposition rates of some organic substrates are reduced. Substantial changes in the species composition of primary producers occur. The richness of phytoplankton species is reduced, while biomass and productivity of phytoplankton are not reduced by acidification. The biomass of herbivorous and predaceous zooplankton is probably reduced because of reductions in numbers of organisms and/or reduction in their average size. Many benthic invertebrates such as species of snails, clams, crayfish, amphipods, and various aquatic insects are intolerant of low pH and are seldom found in acidic lakes. However, certain large aquatic insects such as water boatmen and gyrinids are very acid tolerant and may become the top predators in some acidified lakes. Acidification of aquatic systems has major effects on fish population.

2 Acidification Effects on High-Altitude Alpine Lakes

2.1 Description of the Study Site

Sampling of 20 high-altitude Alpine lakes for analysis of surface water chemistry started in 1980 and occurred usually twice a year at the beginning of summer (mid June to mid July) and in autumn (mid September to beginning of November), both after overturn. The study area is located in the Central Alps in the northern part of Canton Ticino, Switzerland (Fig. 3). In order to study the influence of transboundary air pollution, the freshwater systems were selected in remote areas far from local pollution sources. Since the study area's lithology is dominated by base-poor rocks especially gneiss, its freshwaters are sensitive to acidification. The lake's watersheds are constituted mainly by bare rocks with vegetation often confined to small areas of Alpine meadow. The selected Alpine lakes are situated between an altitude of 1,690 m and 2,580 m and are characterized by intensive irradiation, a short vegetation period, a long period of ice coverage and by low nutrient concentrations. From 2006 the alkaline Lago Bianco was also added to the monitoring list. The location and the main morphometric parameters of the monitored lakes are described in Table 1.

2.2 Average Surface Water Chemistry of 20 High-Altitude Alpine Lakes in the Period 2000–2004

Table 2 shows the mean values of the main chemical parameters measured between 2000 and 2004. Dependent on the lake and the parameter the sample number varies between 7 and 10 (data not shown). The chemical water composition is typical for carbonate-poor mountain regions: low conductivity, alkalinity and pH and small nutrient and DOC (dissolved organic carbon) concentrations. Data of DOC are not shown, they vary between 0.14 and 0.70 mg C l^{-1}. Ortho-phosphate is always smaller than the quantification limit of 4.3 P μg l^{-1} and ammonia and nitrite are negligibly small compared to nitrate (data not shown).

The chemistry of Alpine lakes is the result of numerous physical, chemical and biological reactions occurring in the catchment. These are influenced by many parameters like deposition, catchment area and slope, watershed surface type (vegetation, bare rock), soil, geology and temperature.

If we assume that calcium and magnesium originate exclusively from carbonate, in areas with no acidification the sum of calcium and magnesium should equal alkalinity (Alk $\cong$ $[HCO_3^-]$).

Therefore, an alkalinity that is smaller than the sum of calcium and magnesium concentrations would indicate a consumption of alkalinity. In our study area the estimated consumption varies from 0.025 to 0.071 meq l^{-1} and is on average

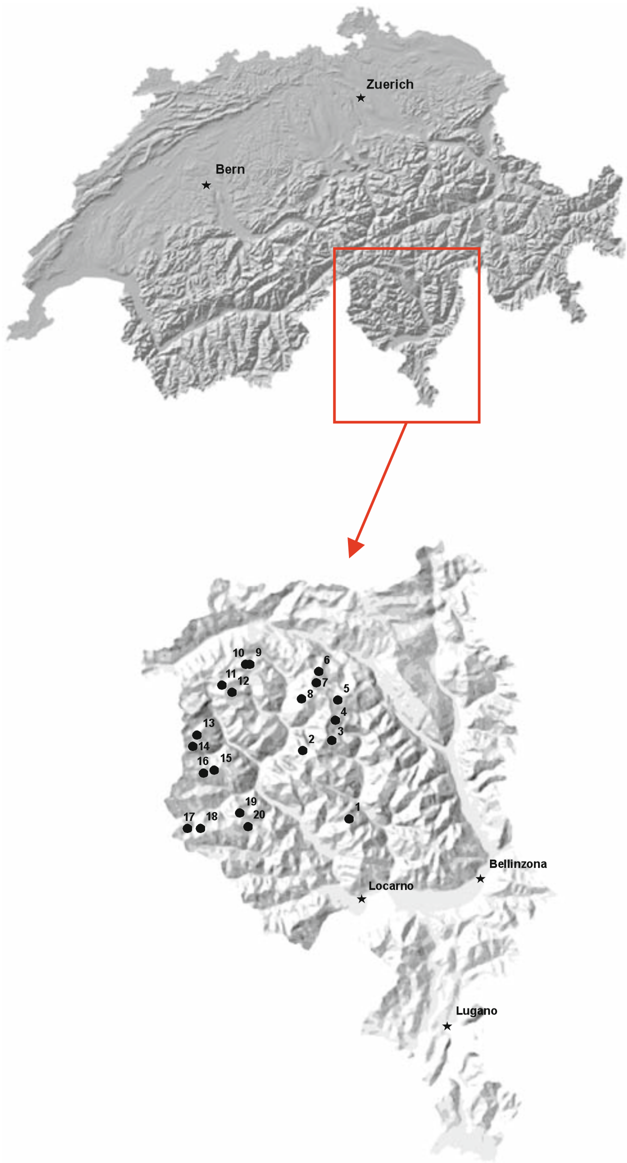

Fig. 3 Monitored lakes (1–21) in Canton Ticino, Switzerland. Copyright: Swiss Federal Office of Topography

Table 1 Location and morphometric parameters of the monitored lakes

Lake number	Lake name	Longitude	Latitude	Altitude m a.s.l.	Catchment area ha	Lake area ha	Max depth m
1	Lago del Starlaresc da Sgiof	8°46′25″	46°16′26″	1,875	23	1.1	6
2	Lago di Tomè	8°41′23″	46°21′47″	1,692	294	5.8	38
3	Lago dei Porchieirsc	8°44′39″	46°22′33″	2,190	43	1.5	7
4	Lago Barone	8°45′06″	46°24′07″	2,391	51	6.6	56
5	Laghetto Gardiscio	8°45′22″	46°45′22″	2,580	12	1.1	10
6	Lago Leit	8°43′17″	46°27′55″	2,260	52	2.7	13
7	Lago di Morghirolo	8°43′00″	46°27′03″	2,264	166	11.9	28
8	Lago di Mognòla	8°41′19″	46°25′49″	2,003	197	5.4	11
9	Laghetto Inferiore	8°35′34″	46°28′34″	2,074	182	5.6	33
10	Laghetto Superiore	8°35′05″	46°28′34″	2,128	125	8.3	29
11	Lago Nero	8°32′22″	46°26′58″	2,387	72	12.7	68
12	Lago Bianco	8°31′10”	46°27′15″	2,077			
13	Lago della Froda	8°33′29″	46°26′24″	2,363	67	2.0	17
14	Laghetto d’Antabia	8°29′32″	46°23′08″	2,189	82	6.8	16
15	Lago della Crosa	8°28′60″	46°22′16″	2,153	194	16.9	70
16	Lago d’Orsalìa	8°31′24″	46°20′23″	2,143	41	2.6	16
17	Schwarzsee	8°30′11′	46°20′10″	2,315	24	0.3	7
18	Laghi dei Pozzöi	8°28′17″	46°15′52″	1,955	33	1.1	4
19	Lago di Sfille	8°29′46″	46°15′52″	1,909	63	2.8	12
20	Lago di Sascòla	8°34′11″	46°17′01″	1,740	90	3.2	5
21	Lago d’Alzasca	8°35′05″	46°15′58″	1,855	110	10.4	40

0.039 meq l^{-1} (Fig. 4a). Similar values were reported by [18]. However, Fig. 4b seems to suggest that the sum of calcium and magnesium concentrations is not completely independent of sulphate concentrations, indicating that carbonate is not the only calcium and magnesium source and that sulphur components can also contribute.

Calcium and/or magnesium sulphates can originate from dry deposition of aerosols, from deposition of dust or from sulphur-containing rocks in the catchment. Since carbonate seems not to be the only calcium and magnesium source, the alkalinity consumption calculated earlier must be considered as an overestimation.

The alkalinity consumption in the lake’s watershed determines its alkalinity and pH. Out of 20, two lakes (Lago del Starlaresc da Sgiof and Laghetto Gardiscio) are acidic (Alk < 0 meq l^{-1}) with average pH’s of 5.2 and 5.3, 13 are sensitive to acidification (Alk < 0.05 meq l^{-1}) and five have low alkalinities (< 0.2 meq l^{-1}) but were not sensitive to acidification. For Lago del Starlaresc da Sgiof it must also be mentioned that the lake is surrounded by a raised bog that may produce acidity itself [20]. Alkalinity and pH correlate closely. It is known, that in open systems, with water in equilibrium with the atmosphere, pH can be represented as a linear function of the logarithm of the bicarbonate concentration [9]:

$$\mathrm{pH} = \log[\mathrm{HCO}_3^-] + 11.3\,(\text{for }25^\circ\mathrm{C}) \tag{4}$$

Table 2 Average surface water concentrations measured between 2000 and 2004 in 20 Alpine lakes. Average values with some or all single values below the quantification limit were preceded with <

Lake name	pH	Cond. (µS cm^{-1})	Alk. (meq l^{-1})	Ca^{2+} (mg l^{-1})	Mg^{2+} (mg l^{-1})	Na^{+} (mg l^{-1})	K^{+} (mg l^{-1})	NH_4^{+} (mg N l^{-1})	SO_4^{2-} (mg l^{-1})	NO_3^{-} (mg N l^{-1})	Cl^{-} (mg l^{-1})	Al dissolved (µg l^{-1})	Al_{tot} (µg l^{-1})
Lago di Tomè	5.6	9.6	0.002	0.93	0.08	0.31	0.14	0.02	1.59	0.47	0.14	44.2	67.5
Lago di Sascòla	6.1	10.1	0.014	0.94	0.15	0.33	0.31	0.02	1.61	0.40	0.15	<23.9	32.5
Lago d'Alzasca	6.8	15.7	0.072	1.88	0.21	0.49	0.43	0.02	2.17	0.27	0.16	<11.3	<14.4
Lago del Starlaresc da Sgiof	5.2	10.4	−0.010	0.67	0.11	0.34	0.19	0.03	1.62	0.42	0.21	88.9	100.9
Lago di Sfille	6.4	9.5	0.025	1.06	0.11	0.37	0.12	0.02	1.59	0.23	0.16	24.7	33.1
Laghi dei Pozzöi	6.5	8.9	0.032	0.96	0.11	0.35	0.16	0.01	1.47	0.12	0.14	18.9	26.1
Lago di Mognòla	6.8	15.7	0.052	1.72	0.24	0.54	0.43	0.01	2.98	0.28	0.12	<12.7	23.7
Laghetto Inferiore	6.5	9.8	0.028	1.08	0.10	0.29	0.32	0.02	1.55	0.27	0.10	<10.5	13.9
Laghetto Superiore	6.4	9.1	0.025	0.96	0.09	0.25	0.28	0.03	1.38	0.26	0.10	<11.4	15.5
Lago d'Orsalìa	6.3	9.6	0.023	1.10	0.08	0.29	0.16	0.13	1.24	0.40	0.12	<13.7	23.1
Lago della Crosa	6.3	7.8	0.017	0.86	0.07	0.26	0.14	0.01	1.08	0.27	0.13	<10.1	<12.8
Laghetto d'Antabia	6.9	13.5	0.067	1.76	0.07	0.42	0.26	0.05	1.25	0.33	0.13	<10.5	<15.9
Lago dei Porchieirsc	6.8	18.3	0.060	2.40	0.14	0.43	0.38	0.02	3.56	0.33	0.13	<11.3	16.3
Lago Leit	6.4	14.1	0.023	1.44	0.25	0.35	0.38	0.02	3.67	0.21	0.11	<12.3	39.3
Lago di Morghirolo	6.6	11.2	0.032	1.17	0.16	0.29	0.36	0.02	2.11	0.21	0.11	<12.1	25.7
Schwarzsee	6.5	11.7	0.038	1.42	0.10	0.31	0.24	0.02	1.54	0.35	0.14	<16.3	24.1
Lago della Froda	6.7	12.2	0.046	1.57	0.10	0.27	0.21	0.01	1.91	0.22	0.09	<13.2	<16.7
Lago Nero	6.8	14.6	0.062	1.87	0.15	0.34	0.38	0.01	2.58	0.16	0.10	<11.1	<14.0
Lago Barone	6.0	9.1	0.012	1.02	0.10	0.26	0.19	0.02	1.75	0.32	0.13	<12.4	22.4
Laghetto Gardiscio	5.3	7.9	−0.006	0.50	0.09	0.18	0.22	0.03	1.62	0.22	0.11	52.9	67.7

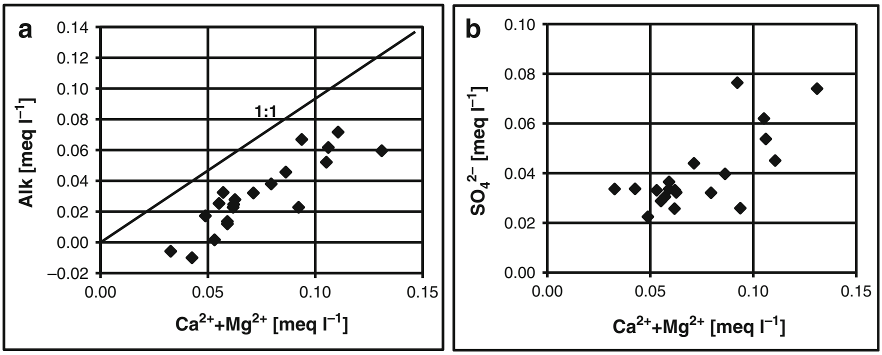

Fig. 4 (**a**) Mean alkalinity and (**b**) SO_4^{2-} concentrations vs. Ca^{2+} + Mg^{2+} concentrations for 20 Alpine lakes

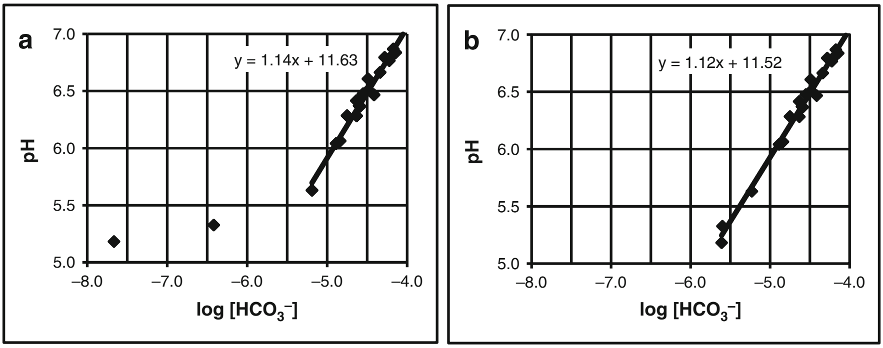

Fig. 5 (**a**) Measured average pH vs. logarithms of bicarbonate (= measured Alk) without consideration of aluminium. (**b**) Measured average pH vs. logarithms of bicarbonate (= measured Alk) added with concentrations of aluminium species for 20 Alpine lakes in the period 2000–2004

Calculating the bicarbonate concentrations with (3), it is possible to plot the measured average pH values of Table 2 against the calculated log[HCO_3^-]. The result is shown in Fig. 5a. For pH >5.5 a linear relationship, very close to that reported in the literature, can be observed (pH = log[HCO_3^-] + 11.2). However, for acid lakes the calculated bicarbonate concentrations seem to be too low. It is reported that at pH<6 the release of metals from soils or sediments as a consequence of weathering processes becomes more and more important. Consequently aluminium hydroxides can influence alkalinity. In [18] the equation for calculating alkalinity was modified as follows:

$$\mathrm{Alk} = [\mathrm{HCO}_3^-] - [\mathrm{H}^+] - 3 \times [\mathrm{Al}^{3+}] - 2 \times [\mathrm{Al(OH)}^{2+}] - [\mathrm{Al(OH)}_2^+] \qquad (5)$$

In Fig. 5b the calculated bicarbonate concentrations of the three lakes with the lowest pH were corrected considering the dissolved aluminium concentrations

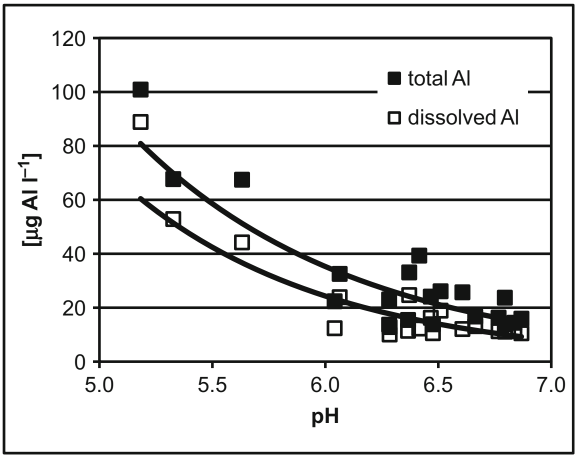

Fig. 6 Mean aluminium concentrations vs. pH for 20 Alpine lakes

specified in (5). The so-obtained points fit very well with the linear regression line, confirming the importance of aluminium in determining alkalinity at low pH's.

The previously mentioned release of aluminium at low pH is particularly important because of its toxic effects on organisms. As expected from theory [9], drawing the average aluminium concentrations against the mean pH a negative exponential correlation is obtained (Fig. 6): Al increases with decreasing pH and lakes with the lowest pH have the highest aluminium concentrations.

The Swiss law for water protection does not indicate a concentration limit for aluminium in surface waters. Therefore, in order to better assess the measured aluminium concentrations, we compared them with the concentration limits existing in the US. The chronic US National Ambient Water Quality Criteria [19] for total aluminium at a pH of 6.5–9 is 87 $\mu g\ l^{-1}$, indicating that the average value over 4 days should not exceed this value more than once every 3 years. However, since aluminium seems to be more toxic at low pH's the result is that in acidic waters the acceptable chronic value of total aluminium may be even lower. We conclude that aluminium concentrations in our three most acid lakes are probably high enough to cause toxic effects on organisms.

2.3 Trends in Surface Water Chemistry of 20 High-Altitude Alpine Lakes Between 1980 and 2004

In order to verify whether atmospheric deposition directly affects surface water chemistry of high-altitude Alpine lakes, trend analyses were performed for 20 mountain lakes with low alkalinity on the key variables involved in acidification and recovery: alkalinity (Gran alkalinity), pH, sulphate, nitrate, base cations (calcium + magnesium). The analysis covers the period 1980–2004 and allows a comparison between atmospheric inputs and surface water quality.

2.3.1 Statistical Methods Used for the Trend Analyses

For each site and parameter we utilized the Mann–Kendall test to detect temporal trends [21]. The two-sided test for the null hypothesis that no trend is present was rejected for p-values below 0.05. In addition we quantified trends with the method of [22]. Results are shown in Table 3.

2.3.2 Sulphate Trends

Sulphate, beside nitrate, is the main acid anion in acid deposition. The trend is significantly negative in 14 lakes. The absence of decreasing sulphate concentrations in the other lakes may be caused by weathering of geogenic sulphur. In fact, in most of these lakes sulphate concentrations are relatively high.

The median sulphate reduction rate is -0.89 μeq l^{-1} yr^{-1}. The decrease of sulphate concentrations started at the beginning of the 1980s, quickly after the beginning of the improvement of the air sulphur oxide concentrations, suggesting a fast chemical response of the lakes. The thin soil and the relatively small sulphur storage, typical for high altitudes, surely contributed to the rapid response in most of the studied lakes.

Similar results were reported in the literature for most European (Scandinavia, UK, Germany, Poland and Czech Republic) and North American (Ontario, Vermont, Québec, Adirondack Mountains, Appalachian Mountains, Blue Ridge Mountains) freshwater sites, where monitoring of the effects of long-range transboundary air pollution on acidification occurs [23].

2.3.3 Nitrate Trends

In any of the 20 lakes a significant nitrate trend can be detected and the median nitrate trend is low (-0.06 μeq l^{-1} yr^{-1}). Interestingly, the mean nitrate trend is almost 15 times lower than the sulphate trend, although the emission reductions of SO_2 (ca. 100 teq yr^{-1}) and NO_x (ca. 90 teq yr^{-1}) in canton Ticino were estimated to be in the same range [24]. However, the fact that in canton Ticino the decrease of SO_2 concentrations in the atmosphere (1.6 μeq m^{-3}) is four times higher than the decrease in NO_x concentrations (0.4 μeq m^{-3}) may indicate that NO_x concentrations in the Southern Alps are influenced by air masses coming from the south (Italy), where reduction of NO_x emissions started later (Switzerland: 1985, Italy: 1992) and is percentually lower (Switzerland: -49%, Italy: -32%) [5]. In addition, those air masses are also enriched with ammonia from livestock and cultivated lands. In fact, from the 1980s to present no trend in nitrogen compounds in rain water has been observed [25]. On the contrary, increasing nitrogen trends were reported for lowland rivers [26] and reveal the presence of increasing nitrogen saturation in soils causing increased nitrate leaching. Transboundary air pollution

Table 3 Results from trend analyses of 20 Alpine lakes monitored between 1980–2004. For each site are indicated the range of the number of data points (depending on the parameter), the calculated trends (Sen's method), the p-value from the Mann–Kendall test and for each parameter the median trend and the number of sites with significant positive or negative or no trend are shown

Lake name	No.	SO_4^{2-} trends		NO_3^- trends		$Ca^{2+} + Mg^{2+}$ trends		Alk trends		H^+ trends	
		μeq l^{-1} yr^{-1}	p-value	μeq l^{-1} yr^{-1}	p-value	μeq l^{-1} yr^{-1}	p-value	μeq l^{-1} yr^{-1}	p-value	μeq l^{-1} yr^{-1}	p-value
Lago di Tomè	14–15	−0.99	0.0008*	0.12	0.5820	−0.24	0.2503	0.25	0.0468*	−0.12	0.6507
Lago di Sascòla	12–15	−1.16	0.0055*	−0.05	0.8820	−0.39	0.1250	0.70	0.0013*	−0.07	0.0093*
Lago d'Alzasca	13–17	−0.94	0.0039*	−0.04	0.7095	0.24	0.5641	1.15	0.0050*	0.00	0.1081
Lago del Starlaresc da Sgiof	13–15	−1.59	0.0003*	0.40	0.4279	−0.22	0.5190	0.25	0.3205	−0.44	0.0238*
Lago di Sfille	12–17	−1.19	0.0002*	−0.04	0.5326	−0.61	0.2165	0.45	0.0625	−0.01	0.3131
Laghi dei Pozzöi	12–16	−1.14	0.0069*	−0.30	0.1251	−0.49	0.1497	0.95	0.0148*	−0.01	0.3291
Lago di Mognòla	13–17	0.13	0.7729	0.00	0.8030	0.33	0.8691	−0.09	0.8040	0.00	0.5030
Laghetto Inferiore	18–22	−1.07	0.0000*	−0.29	0.1739	−0.33	0.3974	0.41	0.5716	0.00	0.3081
Laghetto Superiore	14–18	−0.98	0.0000*	−0.14	0.2532	0.07	0.7909	1.11	0.0044*	−0.02	0.2351
Lago d'Orsalìa	12–16	−1.09	0.0307*	0.24	0.3905	0.41	0.3219	1.21	0.0003*	−0.07	0.0028*
Lago della Crosa	13–16	−0.83	0.0000*	−0.02	0.4605	−0.15	0.3439	0.83	0.0002*	−0.03	0.1207
Laghetto d'Antabia	10–13	−0.63	0.0015*	0.02	0.8055	0.90	0.0381*	1.57	0.0031*	0.00	0.1381
Lago dei Porchieirsc	9–11	0.06	0.7548	−0.14	0.3850	0.84	0.9379	1.20	0.0423*	−0.02	0.0112*
Lago Leit	12–14	1.14	0.0004*	−0.16	0.0990	1.82	0.0002*	0.70	0.0010*	−0.03	0.0018*
Lago di Morghirolo	13–16	−0.12	0.4713	−0.17	0.1592	0.52	0.0132*	0.93	0.0039*	−0.02	0.0334*
Schwarzsee	12–16	−1.02	0.0307*	−0.10	0.6842	−0.38	0.6525	0.97	0.0581	−0.01	0.2553
Lago della Froda	12–16	−0.64	0.0272*	−0.18	0.1140	0.14	0.7187	1.09	0.0039*	−0.01	0.0123*
Lago Nero	13–18	−0.07	0.5442	−0.06	0.3395	0.09	0.7332	0.84	0.0056*	0.00	0.7015
Lago Barone	14–16	−0.61	0.0954	0.02	0.7856	0.45	0.1258	0.84	0.0090*	−0.05	0.2019
Laghetto Gardiscio	13–16	−0.39	0.0169*	−0.13	0.2963	0.07	0.7187	0.25	0.0909	−0.29	0.0255*
n° increasing trend		1		0		3		14		0	
no trend		5		20		17		6		12	
n° decreasing trend		14		0		0		0		8	
median		−0.89		−0.06		0.08		0.84		−0.02	

*$p<0.05$

may therefore explain the difference between trends of Swiss nitrogen emission into the atmosphere and nitrate concentrations in Alpine lakes.

For other European and North American sites, where monitoring of the effects of long-range transboundary air pollution on acidification occurs, a general trend in nitrate concentrations could also not be detected. Significant negative trends were only found in North America in the Adirondack, Appalachian and Blue Ridge Mountains. At other North American (Maine, Atlantic Canada, Ontario, Vermont and Quebec) and European (UK, Germany, Poland, Czech Republic, Scandinavia) sites nitrate trends were not significant but concentrations seemed to mainly decline [23].

2.3.4 Base Cation Trends

Base cations are mobilized by weathering and cation exchange reactions that neutralize acids in the watershed. They respond therefore indirectly to changes in sulphate and nitrate concentrations. In fact, if acid anion concentrations (mainly sulphate) decrease, base cations are also expected to decrease. However, in the last few years an increase of the occurrence of alkaline rain episodes (probably due to climatic effects) has been observed and it is likely that calcareous Saharan dust, rich in base cations, is responsible for it [27]. Accelerated weathering, resulting from recent climate warming may also contribute to higher base cation concentrations [28].

As usually done for similar studies [23, 29], concentrations of calcium and magnesium are utilized as a surrogate for non-marine total base cation concentrations, because these cations are quantitatively most important.

For Ca^{2+}+ Mg^{2+} no regional trend can be observed. At our study site only three lakes out of 20 show a significant increasing trend. The median rate is low: 0.08 μeq l^{-1} yr^{-1}. The absence of a significant base cation trend in most lakes may be explained by the fact that base cations are controlled by both concentration increasing and decreasing mechanisms.

Similar results were observed in most European countries (UK, Germany, Poland, Czech Republic) and North American sites (Adirondack mountains, Blue ridge mountains, Maine, Atlantic Canada, Ontario). Significant trends were only found in Scandinavian countries and in the Appalachians Mountains, Vermont and Quebec. However, independently of the presence or not of a significant trend, at most studied sites concentrations of base cations tend to decrease, phenomena that cannot be observed in the Alps [23].

2.3.5 Alkalinity Trends

When rates of Ca^{2+}+ Mg^{2+} decline are equal, or nearly equal, to rates of sulphate and nitrate reduction, then recovery (increasing alkalinity and pH) is prevented (see Eq. 2). Subtracting the median acid anion trends (−0.89 and −0.06 μeq l^{-1} yr^{-1}) from the median base cation trend (0.08 μeq l^{-1} yr^{-1}) we obtain a

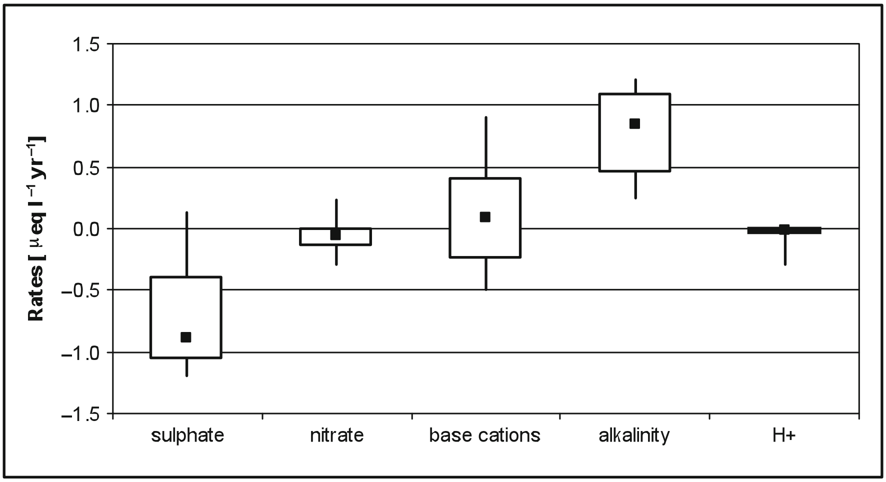

Fig. 7 Distribution of sulphate, nitrate, base cation and H^+ rates of the trends calculated for the study site. Each *box* shows the range of the trends (25th to 75th percentile, with *dot* at median). The confidence intervals indicate 10th and 90th percentiles

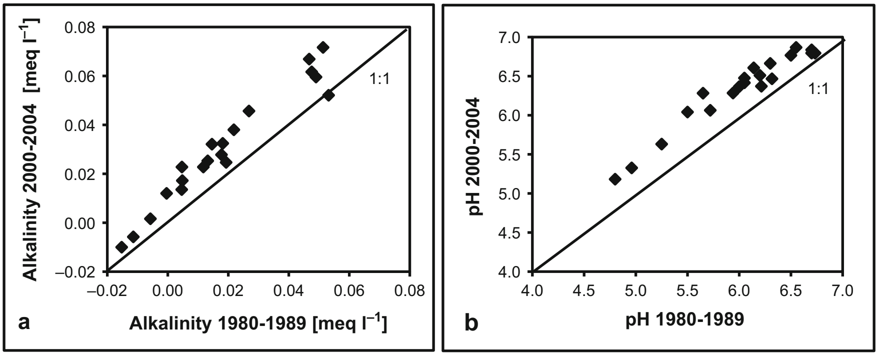

Fig. 8 (**a**) Average alkalinity between 2000 and 2004 vs. average alkalinity in the period 1980–1989. (**b**) Average pH between 2000 and 2004 vs. average pH in the period 1980–1989 in 20 Alpine lakes

figure (1.03 μeq l^{-1} yr^{-1}) that suggests a positive alkalinity trend at our study site (see also Fig. 7). In fact, we can find a significant positive alkalinity trend in most lakes (14).

In Fig. 8a average alkalinity values in the 1980s are compared with average values between 2000 and 2004. It appears that, although not in all lakes a significant positive alkalinity trend can be detected, in almost all lakes an alkalinity increase from the 1980s to present can be observed. The average increase is 0.012 meq l^{-1}. In the 1980s four lakes were acidic (today two), 14 were sensitive to acidification (today 13) and two had low alkalinities but were not acid sensitive (today five). In terms of critical load exceedances it can be observed that today six lakes have an average alkalinity that is below the critical value (0.020 meq l^{-1}) while there were 13 in the 1980s.

A general increase in alkalinity was also observed at most North American and European sites, where monitoring of the effects of long-range transboundary air pollution on acidification occurs. The largest increase occurred in Southern Scandinavia (2.3 μeq l^{-1} yr^{-1}) and in western central Europe (western Germany, 3.5 μeq l^{-1} yr^{-1}). No increase could be observed in the UK, Maine, Atlantic Canada and Ontario. It has been observed that at DOC-rich sites, a decrease in acid deposition could also cause an increase in concentrations of DOC and therefore organic acidity, which could counteract the increase of alkalinity [23]. A typical example of this phenomenon are sites in the UK. Increasing DOC concentrations and the influence of sea-salt episodes inhibited an increase of alkalinity, although increase of pH and decrease of aluminium concentrations were evident. [23]

2.3.6 H^+ Trends

The median H^+ trend is low (−0.02 μeq l^{-1} yr^{-1}) and only eight lakes have a significant decreasing trend. Plotting the calculated H^+ trend against the average H^+ concentrations during the whole monitoring period a correlation between the two variables appears (Fig. 9): higher H^+ decrease at higher H^+ concentrations or in other words higher H^+ decrease at lower pH. This phenomena is probably caused by the high buffer capacity of bicarbonate-containing water around pH = 6.3 (5.8–6.8). This implies that in the pH range ≈4.3–6.3, the same sulphate decrease generates a higher H^+ decrease at lower than at higher pH.

The pH improvement in the studied lakes becomes evident when comparing the average pH in the 1980s with those of the years 2000–2004 (Fig. 8b). The increase

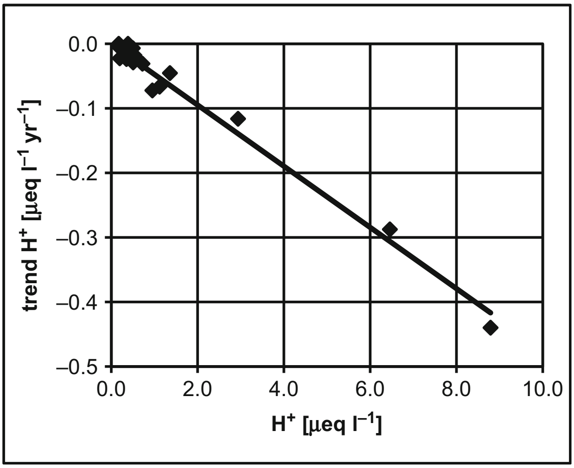

Fig. 9 H^+ trends vs. average H^+ concentrations during the period 1980–2004 for 20 Alpine lakes

of pH was on average 0.3. In the 1980s eight out of 20 lakes had an average pH < 6, while presently only three lakes have pH values that are critical for aluminium dissolution.

Even better recovery has been observed for other European countries, participating in the cooperative programme on assessment and monitoring of acidification of rivers and lakes. Trends were significant at most sites and the largest changes in pH occurred in the UK (-0.54 μeq l^{-1} yr^{-1}). In North America changes were only significant in Maine and Atlantic Canada. No significant trends were found in Vermont, Québec, Ontario, Adirondack Mountains, Appalachian Mountains and Blue ridge Mountains.

To better interpret the observed pH improvement it would be useful to know pH value's before lake acidification began. As far as we know, for the monitored lakes, pH reconstructions exist only for Lago Leit. [30] estimated from the Chrysophycean scale a pH decrease from ca. 6.8 to 5.6 (pH decrease = 1.2). However, our pH measurements indicate that in Lago Leit pH generally did not decrease below pH 6, indicating that the pH decrease estimated by [30] is probably slightly overestimated. [18] reconstructed the pH's of another two Alpine lakes, that are not included in our monitoring list, from diatom scales (Langsee and Lago della Zòta). The pH of Langsee probably ranged between 5.7 and 5.9 until 1900 and decreased to 5.1 (1990–1991) in the twentieth century. Similar results were obtained for Lago della Zòta: pH's of 5.7–5.8 before 1900 and a decrease to 5.0 (1991). [28] applied the dynamic model MAGIC on an Italian mountain lake (Paione Superiore). The modelled pH drop was 1.5. Although it is clear that the pH decrease in the twentieth century as a consequence of acid deposition is specific for each lake, comparing our observed pH increase in the last 20 years with literature regarding the pH trend in the last century, it can be concluded that acid-sensitive high-altitude Alpine lakes had still not completely recovered.

Interestingly the equation of the linear regression in Fig. 9 corresponds to the differential equation

$$\mathrm{d}[\mathrm{H}^+]/\mathrm{dt} = -0.0474 \times [\mathrm{H}^+](\mathrm{t})$$

which is the derivate of

$$[\mathrm{H}^+](\mathrm{t}) = [\mathrm{H}^+](\mathrm{t}=0) \times \mathrm{EXP}(-0.0474 \times \mathrm{t})$$

describing the concentration of protons as a function of time during the studied period (1980–2004). This equation permits us to calculate for the three lakes with a pH still below 6 the time still necessary to reach the critical value for aluminium mobilization (=pH 6), assuming that reduction of the atmospheric load of acidity proceeds with the same rate as observed during 1980–2000 also after 2000. To reach this goal for Lago di Tomè (ΔpH = 0.4) another 18 years are necessary while for Lago del Starlaresc da Sgiof and Laghetto Gardiscio (ΔpH = 0.7–0.8) still 40 years have to pass. However, it is clear from the actual emissions of sulphur and nitrogen into the atmosphere (Fig. 1) that the atmospheric load of acidity doesn't decrease further at the same rate as observed for 1980–2000. In order to obtain that, a dramatic reduction of NO_x emission must occur. However, even if this would happen, the latest around 2025–2030 the load of acidity would fall below zero, indicating that at least the two most acidic lakes Starlaresc da Sgiof and Laghetto Gardiscio have probably never had and never will have a pH above 6.

2.4 Acidification Effects on the Population of Macroinvertebrates of Five High-Altitude Alpine Lakes

The ultimate goal of emission control programmes is biological recovery, for example the return of acid-sensitive species that have disappeared and the restoration of biological functions that have been impaired during the course of acidification. To study the effect of acidification on biology, macroinvertebrates were utilized as bioindicators. The response of macroinvertebrates to water acidity is often precisely the reverse of their indicator properties for effluent pollution. A range of oxygen-dependent cold-water organisms which react very sensitively to pollution from effluents are astonishingly tolerant to low pH values. Since 2000 macroinvertebrates have been collected by "kick sampling" in the littoral and in the outlet of four high-altitude Alpine lakes (Laghetto Inferiore, Laghetto Superiore, Lago di Tomè, Lago del Starlaresc da Sgiof) usually two to three times a year. In order to compare the macroinvertebrate population of acid and acid-sensitive lakes with an alkaline lake, Lago Bianco was added to the monitoring list in 2006. Data from 2005 and 2006 are presented below.

In order to determine the "biological health" of surface waters with respect to acidification different approaches were used. For all samples the total E.B.I. (= Extended Biotic Index) taxa number according [31] and the EPT (= number of families from the orders Ephemeroptera, Plecoptera, Trichoptera) index was calculated. Both the taxa richness and the EPT index are indicators for the "health" of a biological community. In particular, the EPT index is often used as a water-quality indicator because macroinvertebrates belonging to the orders of Ephemeroptera, Plecoptera and Trichoptera are highly sensitive to pollution. In addition, for Alpine rivers the German classification system of [32] has been shown to be a valuable method [33]. This categorisation system permits one to evaluate and assess the acidity of rivers on the basis of macroinvertebrate populations. Unfortunately, for high-altitude lakes, because of their natural poorness in taxa, there still does not exist a viable macroinvertebrate classification method that is able to describe water acidity. However, it is possible to evaluate the composition of macroinvertebrate populations with regard to acid sensitiveness by applying indexes from the acid-classification systems of [32] to single taxa and omitting to attribute a specific acidification category to the entire sample.

Results are summarized in Table 4. The table reports for each year and lake the number of samples per year, the number of organisms, the number of E.B.I. taxa,

Table 4 Number of samples, organisms, taxa, EPT index, number of taxa for each Braukmann and Biss (B&B) index and pH in the emissary and of five Alpine lakes during 2005 and 2006. LI, LS, LT, LSt, LB represent Laghetto Inferiore, Laghetto Superiore, Lago di Tomè, Lago del Starlaresc da Sgiof, Lago Bianco

	LI		LS		LT		LSt		LB
	2005	2006	2005	2006	2005	2006	2005	2006	2006
No. of samples	3	2	3	2	2	2	2	2	2
No. organisms	8338	6086	6631	5742	2160	3066	2730	6293	6195
No. taxa E.B.I.	21	20	18	17	15	17	16	15	19
EPT index	11	9	8	9	5	6	3	4	7
No. taxa with B&B index = 1	0	0	0	0	0	0	0	0	0
No. taxa with B&B index = 2	4	4	3	3	0	0	0	0	5
No. taxa with B&B index = 3	2	2	1	1	0	0	0	0	1
No. taxa with B&B index = 4	4	4	3	3	4	5	1	1	2
No. taxa with B&B index = 5	2	2	3	2	3	3	1	2	1
pH	6.5	6.3	6.5	6.5	5.7	5.5	5.3	5.4	7.6

the EPT index and also the number of taxa for the five “Braukmann and Biss” indexes, whereas the smallest index refers to the most acid-sensitive taxa. In order to better interpret results average pH’s are also indicated.

In general, lake acidity seems to influence the population of macroinvertebrates. In fact, the higher pH’s of Lago Bianco, Laghetto Inferiore and Laghetto Superiore compared to Lago di Tomè and Lago del Starlaresc da Sgiof seem to become reflected in a higher taxa richness, EPT index and the presence of organisms with lower “Braukmann and Biss indexes”. Important differences regarding the macroinvertebrate population between the alkaline Lago Bianco and the low acid lakes (Laghetto Inferiore, Laghetto Superiore) are not observed. It seems that biological recovery as a consequence of chemical recovery cannot be investigated for high-altitude Alpine lakes as the available biological data series (from 2000) is not as long as the chemical dataset (from the 1980s). Differently, signs of biological recovery have been described for numerous sites in Canada, Norway, Sweden and UK, where the reappearance of acid-sensitive species has been observed [23]. It is probable that, although not monitored, a similar phenomena has occurred in the Alps as well.

2.5 *Discussion and Conclusions*

Trend analysis over the last 25 years of the main chemical parameters involved in acidification processes revealed, in agreement with the overall observed SO_2 emission reduction, significant decreasing sulphate and increasing alkalinity trends in most studied high-altitude Alpine lakes. Differently, trends for base cation and nitrate were small and mostly insignificant. The average increase in alkalinity between 1980 and 2004 was 0.012 meq l^{-1}, while the average pH increase during the same time period was 0.3. pH reconstructions mentioned in the literature suggest that complete recovery has still not occurred.

At present two of 20 studied high Alpine lakes are still acid (Alk <0 meq l^{-1}), 13 are sensitive to acidification (0 <Alk <0.05 meq l^{-1}) and five have low alkalinities but are not acid sensitive (0.05 meq l^{-1} < Alk <0.2 meq l^{-1}). In the 1980s the number of acid, acid sensitive, and not acid sensitive lakes was 4, 14 and 2, respectively. Because of the increase in aluminium dissolution a pH decrease below 6 can become critical for the biology of lakes. Today only three lakes from 20 have an average pH below 6. In the 1980s there were seven.

A pH below 6 changes significantly the macroinvertebrate population. In fact, the presently higher pH of Lago Bianco (7.6), Laghetto Inferiore (6.4) and Laghetto Superiore (6.5) compared to Lago di Tomè (5.6) and Lago del Starlaresc da Sgiof (5.3) becomes reflected in a higher taxa richness, EPT index and the presence of organisms with lower "Braukmann and Biss indexes" in emissary samples.

Concerning future trends [29] writes:

> "the uncertainties in future chemical recovery mainly relate to the effects of climate change and the future behaviour of nitrogen in the ecosystem".

Because of decreased SO_2 emissions, the importance of nitrogen for freshwater acidification becomes more and more determinant. Therefore, uncertainties in the evolution of nitrogen emissions make it difficult to predict future trends in freshwater recovery. On the other hand, climate warming may increase the number of alkaline rain episodes (containing Saharan dust) and enhance weathering reactions, which buffer better acid precipitation, increasing alkalinity and pH.

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Alpine Glaciers as Archives of Atmospheric Deposition

M. Schwikowski and A. Eichler

Abstract Alpine glaciers are natural archives of past precipitation. At high elevations where melting is negligible and precipitation occurs as snow throughout the year the manifold information contained in the annual snow layers is well preserved. This information is accessed by ice core drilling and analyses. The stable isotope composition of water and the chemical impurities in the ice allow reconstructing past climate conditions and air pollution. The time period covered by suitable glaciers in the Alps depends on accumulation rate and glacier thickness and varies from a few hundred to more than 10,000 years. Concentration records of various trace species demonstrate the impact of anthropogenic emissions on the impurity content of snow. They show a generally consistent picture of a vastly altered atmospheric composition due to anthropogenic activities. Compared to rain samples from the vicinity of Zurich, concentrations of chemical impurities in Alpine ice are lower by a factor of 3–4, reflecting the different vertical and horizontal distance to the emission sources.

Keywords Air pollution, Alpine glaciers, Climate reconstruction, Ice core, Natural archives

Contents

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1 Glaciers as Natural Archives

Alpine glaciers are important fresh water reservoirs. In Switzerland the total numbers of glaciers is 5,422, covering an area of 3,010 km^2 [1], and forming a storage of 45 million m^3 which is about 19% of the entire fresh water reservoir [2]. In addition to this invaluable function as a fresh water source, alpine glaciers serve as natural archives of past environmental conditions. This is due to the fact that glaciers form by continuous accumulation of snow over time, when cool temperatures and sufficient snow precipitation prevail. These conditions found for instance at high altitudes in mountain areas preserve annually deposited snow. Only a few hours after deposition snow will begin to undergo metamorphism driven by temperature gradients. Snow crystals become progressively larger and more rounded over time. Under the pressure of the layers of snow on top, this granular snow fuses into denser firn. Over a period of years, the firn undergoes further compaction and slowly converts to ice allowing the formation of glaciers. Thus, glaciers consist of layers of snow deposited year by year (Fig. 1) and are therefore precious archives of precipitation in the past.

However, in the Alps there are only a few glaciers suitable as natural archives. A prerequisite is that firn temperatures are well below 0°C and melt-water percolation is negligible in order to preserve the signal of past climate and environmental changes. Such conditions can be found above 4,000 m a.s.l. in the Northern part and above 4,300 m a.s.l. in the Southern part of the Alps [3]. Thus, potential sites for obtaining useful samples are limited to a few high-elevation areas, for instance the Bernese Alps, the Monte Rosa area, and the Mont Blanc area.

2 Ice Core Drilling, Preparation, and Analyses

Samples from the glacier archives are collected by electromechanical ice core drilling. Ice core sections are obtained as cylinders with for instance 8-cm diameter and 70-cm length. For the typical thicknesses of suitable high-alpine glaciers, i.e. 50–200 m, drilling is conducted in a dry borehole. Only for deeper drilling (>300 m) as necessary on the polar ice sheets, the borehole has to be stabilized by introducing drilling fluid. Ice core sections are transported in frozen condition to the laboratory. In a cold room at −20°C they are prepared for chemical analysis by cutting smaller samples with an electrical band saw. A wide range of different parameters are analyzed in the ice cores using various instrumental analysis techniques. For example, stable isotope ratios ($^{16}O/^{18}O$, $^{1}H/^{2}H$) are determined with isotope ratio mass spectrometry [4], concentrations of trace

Fig. 1 Climbers ascending Weissmies (4,023 m) in the Swiss Alps. The crevasse wall clearly shows the layering of snow. The most prominent layers confine the accumulation of individual years. The yellowish layer is a Saharan dust deposit, a phenomenon commonly occurring in the Alps. Photo Jürg Alean, Glaciers online: http://www.glaciers-online.net

species with ion chromatography [5] (Fig. 2) or ICP-mass spectrometry [6], radio nuclides (^{210}Pb, ^{3}H, etc.) with α-spectroscopy or liquid scintillation counting [7], organic and elemental carbon with a thermal method [8], pesticides with GC mass spectrometry [9], and cosmogenic nuclides (^{14}C) with accelerator mass spectrometry [10].

Most of the chemical analyses are performed after melting of the snow samples which subdivides the species into water-soluble and insoluble ones. The concentrations of trace species can vary by at least two orders of magnitude and the lowest values are in the range of 1–10 μg/l for major ions such as Ca^{2+}, NH_4^+, NO_3^-, SO_4^{2-}, and 10–100 ng/l for trace metals. Thus, the snow and ice samples are particularly sensitive with respect to contamination and require special care in sample handling.

3 Information Contained and Time Periods

The information contained in the glacier archive is manifold. The composition of the water molecules (stable isotope ratio of hydrogen $^{1}H/^{2}H$ and oxygen $^{16}O/^{18}O$) is often used as a paleo thermometer, for example [4]. In addition, the ice contains

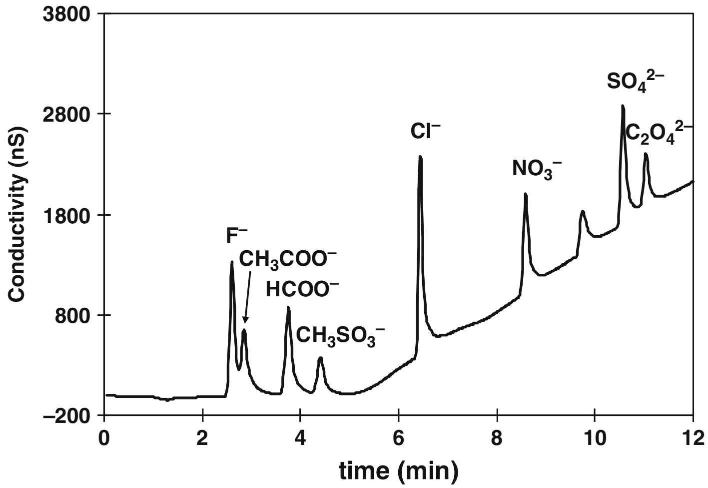

Fig. 2 Ion chromatogram of a sample of melted Alpine ice for direct simultaneous determination of inorganic (F^-, Cl^-, NO_3^-, SO_4^{2-}) and organic anions (CH_3COO^-, $HCOO^-$, $CH_3SO_3^-$, $C_2O_4^{2-}$)

trace species scavenged from the atmosphere as aerosol particles and/or trace gases during snow formation. Such trace species can originate from natural (volcanic eruptions, sea spray, mineral dust) and anthropogenic sources (fossil fuel combustion, energy production, traffic, agriculture) and the concentration in ice is assumed to reflect air pollution in the past.

The time period covered by an ice core depends on snow accumulation rate, glacier thickness, and ice flow characteristics. Mean annual precipitation in the Alps varies between 0.5 and 3 m. Accordingly, relatively high annual net accumulation rates were observed at most of the ice core sites, i.e. ~1.7 m water equivalent (weq) at Fiescherhorn glacier, Bernese Alps, ~1.5 m weq at Col du Dôme, Mont Blanc, ~1.6 m weq at Lys Glacier, ~2.7 m weq at Grenzgletscher, both Monte Rosa area, and ~2.6 m weq at Piz Zupó, Bernina area. A noteworthy exception is the Colle Gnifetti glacier saddle in the Monte Rosa area on the border between Switzerland and Italy. Because of the preferential wind erosion of dry winter snow the annual snow accumulation is in the order of 0.3 m weq only [11]. Ice cores are drilled in the upper zones of a glacier, the so-called accumulation area, where the glacier mass balance is positive (Fig. 3). Here the glacier would become continuously thicker if there were not a compensating flow of ice to the so-called ablation zone. There the glacier has a negative mass balance due to melting. Glacier flow is a simple consequence of the properties of ice, undergoing plastic deformation due to gravity and weight. As a result, annual layers become thinner with increasing depth. Thinning rates are particularly high in cold glaciers where the ice is frozen to the bedrock. The resulting strongly nonlinear age–depth relationship is illustrated in Fig. 4 on the examples Fiescherhorn glacier and

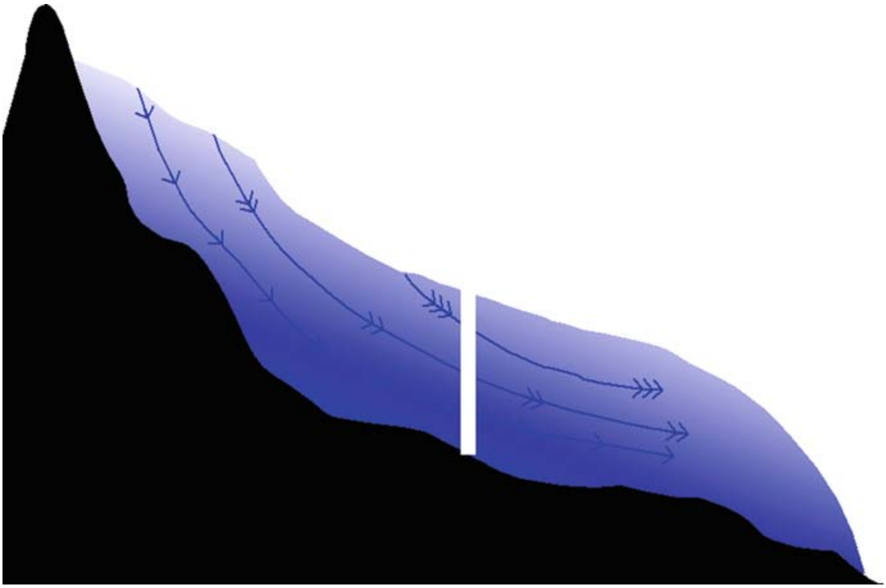

Fig. 3 Schematic of a high-alpine glacier with location of the borehole. *Lines* show the flow direction of individual ice particles, whereas the velocity is indicated by the number of *arrow heads*. Courtesy of Aurel Schwerzmann

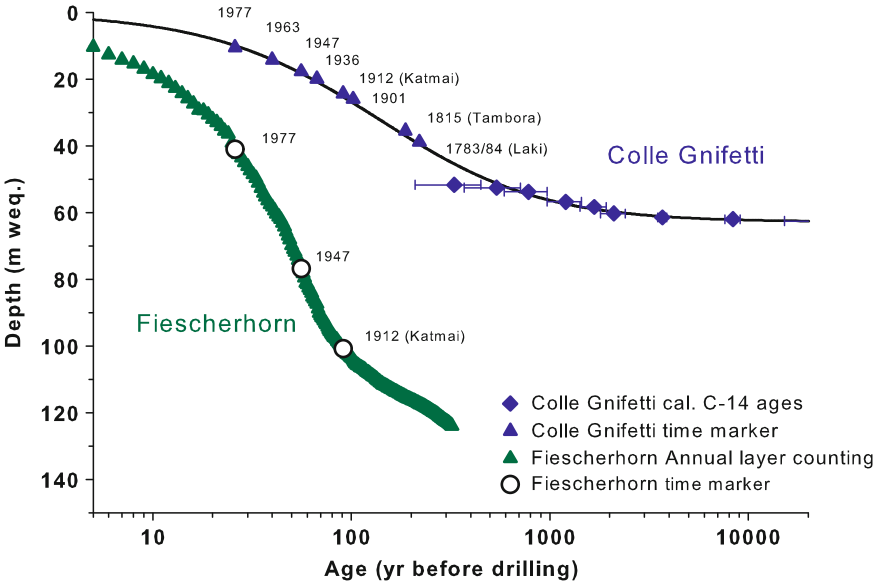

Fig. 4 Depth–age relationship of the ice cores from Fiescherhorn glacier [12] and Colle Gnifetti [13, 14]. Besides annual layer counting and radiocarbon (^{14}C) dating, two types of time markers were used: Saharan dust events (labeled by the year only) and volcanic eruptions (labeled by year and name of volcano). Depth is given in water equivalent. This is the amount of water contained in the ice core which is calculated using firn and ice density, respectively, both increasing with depth

Colle Gnifetti. Fiescherhorn glacier has a high accumulation rate of 1.7 m weq and is not frozen to the bedrock. Thus, the 150-m long ice core (124 m weq) covers only the last 340 years [12]. Colle Gnifetti has a much lower accumulation rate and is frozen to the bedrock. Although its thickness is only 80 m, ice over 10,000 years old was found in the deepest part of the core [13]. The dating of ice cores is performed by annual layer counting using one or a combination of more than one seasonally varying signal (such as stable isotope ratio, ammonium or dust concentration), identification of reference horizons (Saharan dust, volcanic layers etc), radiocarbon dating using carbonaceous aerosols, and glacier flow modeling [14].

4 Chemical Components Scavenged by Snow and Preserved in Ice

The chemical composition of snow is assumed to reflect the concentrations of the various impurities present in the atmosphere at the time of snow formation. These can be either species contained in atmospheric aerosol particles (e.g. ammonium, calcium, hydrogen ion, nitrate, sulfate, soot, organic components, insoluble minerals, trace metals) or reactive trace gases (e.g. hydrochloric acid, nitric acid, formaldehyde, hydrogen peroxide). Nonreactive trace gases such as carbon dioxide, methane, and nitrous oxide are not captured by snow crystals and can be determined in air bubbles formed in glacier ice, for example [15]. They will not be discussed in this context, since only preliminary work has been carried out on samples from the Alps so far. An important aspect of Alpine ice cores is their proximity to emission sources of anthropogenic pollution. Especially for aerosol particles and related gaseous species, with short atmospheric life-times, changes in the atmospheric load due to human activities and their possible impact on climate are most pronounced in industrialized areas. Detailed studies of the atmospheric transport from the emission source areas to the high altitudes, where the glacier archives are located, have been conducted. They show that the high-altitude aerosol concentration and subsequently also the concentration of aerosol constituents in snow and ice is strongly controlled by the seasonally varying intensity in vertical mixing, see for example [16]. Seasonal amplitudes are large compared to the long-term concentration trends and show pronounced interannual variability. Thus, to obtain long-term trends in the aerosol concentrations averaging over longer time periods (5–10 years) is required.

5 Long-Term Concentration Trends of Selected Species

Various concentration records of a number of chemical trace species and gases obtained from the different Alpine ice cores have been published, see for example [17–22]. These records clearly demonstrate the impact of anthropogenic emissions on the impurity content of snow. They show a generally consistent picture of a vastly altered atmospheric composition due to industrialization. This is the case for major aerosol components such as sulfate, nitrate, ammonium, and carbonaceous particles, but also the concentrations of trace substances, for example heavy metals, fluoride, chloride, and radionuclides are affected. Furthermore, the presence of organochlorine pesticides in Alpine ice could be detected [9].

Most of the pollutants have their highest concentrations in the 1970s and early 1980s due to a maximum of anthropogenic emissions. After that time a downward trend of the concentrations shows the response to air pollution control measures in Europe (such as introduction of filters, catalytic converters, unleaded fuel) as illustrated in Fig. 5 on the examples of sulfate, lead, and black carbon. Remarkable exceptions are nitrate, see for example [22] and ammonium, which are still at a high level. In the case of nitrate, reductions of NO_x emissions by individual cars have been counterbalanced by increasing traffic. Ammonium originates from agricultural activities such as animal manure and fertilizer application, where no control measures have been introduced.

Ice core-derived concentration records of aerosol species have been used to estimate the aerosol effect on climate. For this purpose, records from mid-latitude glaciers are extremely important. On the one hand, only few long-term records of tropospheric aerosols exist, and on the other hand, data from the emission source areas are needed because of the regional nature of aerosol concentrations. Sulfate concentrations in ice cores from the Alps display the strongest anthropogenic influence of the major aerosol constituent during the last 150 years, with a concentration increase by a factor of 13 between the preindustrial and industrial period. The level in the Alps in the period of maximum industrial emissions is about a factor of five higher than in Greenland ice cores, whereas in Antarctica no anthropogenic sulfate could be detected [23].

6 Comparison of Concentration Levels in Alpine Ice and in Precipitation at Lower Elevation

Monitoring of the composition of rain water in Switzerland began in 1985 at five different sites in the frame of the National Air Pollution Monitoring Network [24]. In Table 1 average concentrations of sulfate, nitrate, and ammonium at the site

Table 1 Average concentrations of sulfate, nitrate, and ammonium in precipitation at Dübendorf, agglomeration Zurich [24], and Colle Gnifetti (summer values only) over the time period 1985–2006

Site	Elevation (m asl)	Sulfate (mgS/L)	Nitrate (mgN/L)	Ammonium (mgN/L)
Dübendorf	430	0.55	0.40	0.55
Colle Gnifetti	4,450	0.12	0.07	0.10

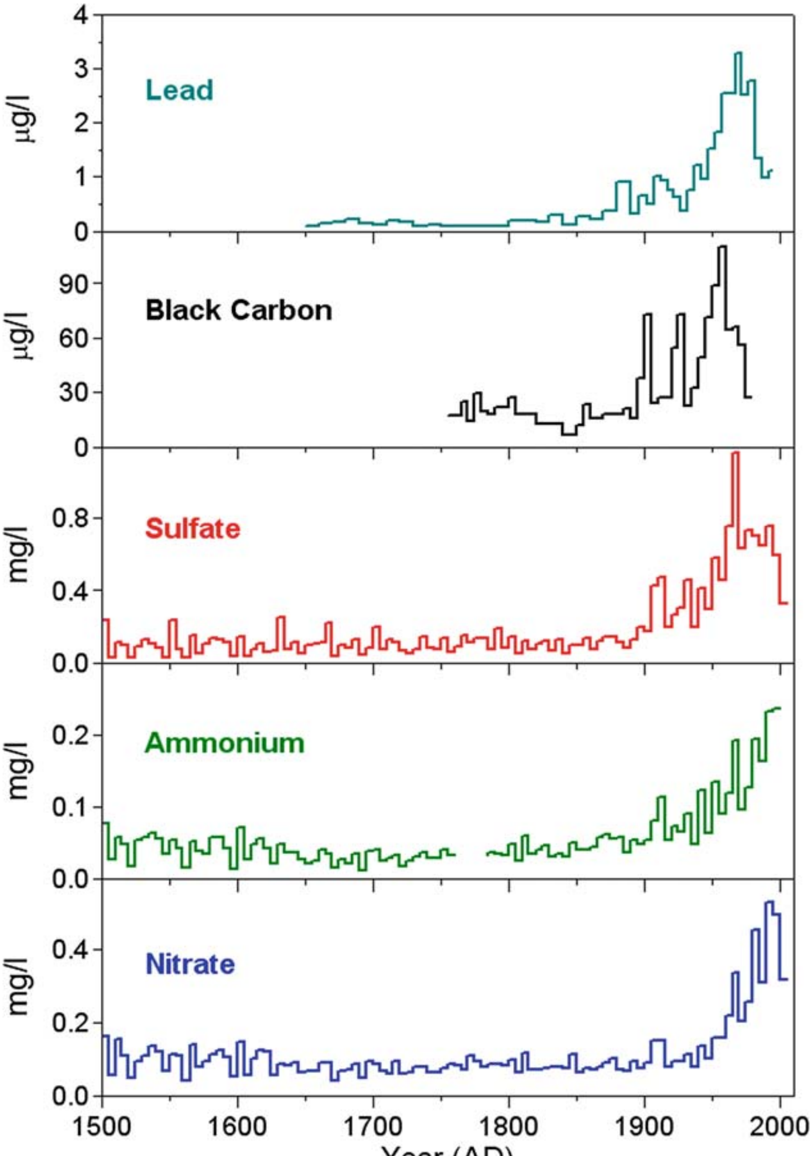

Fig. 5 Concentration records of different trace species in the ice core from Colle Gnifetti glacier as 5-year averages

Dübendorf, belonging to the agglomeration Zurich, and at the glacier Colle Gnifetti are compared. Concentrations in the Alpine ice are lower by about a factor of 3–4 than in rain at Dübendorf. Generally, this reflects the strong vertical aerosol concentration gradient in the atmosphere with highest values close to the emission sources of the precursor gases located at lower elevations. At the typical altitude of Alpine glaciers air masses from the polluted boundary layer are diluted with cleaner free tropospheric air. In addition, during precipitation events, removal of air pollution by precipitation scavenging well below the altitude of Alpine glaciers prevents polluted air masses reaching the glacier sites.

Concentrations in precipitation at the low and high elevation sites show different temporal trends over the time period 1985–2006 as illustrated on the example of sulfate in Fig. 6. A clear downward trend of sulfate in rain is observed at the lower site Dübendorf, whereas in the Alpine snow and ice the trend is weaker. A similar absence of a strong trend is reported for aerosol concentrations at the high-alpine site Jungfraujoch (3,550 m asl, [25]). The distinct behavior is explained by the different catchment areas of low and high-elevation sites. Low-elevation sites receive air masses from the vicinity and concentrations are representative of local to regional conditions, i.e. in the case of Dübendorf the downward trend in sulfate reflects air pollution control measures in Switzerland. In contrast, high-elevation sites are influenced by emissions of much larger source areas. Using source-receptor calculations in order to allocate the sources of air pollution arriving over the Alps, Spain, Italy, France, and Germany were found to be the main contributors [19]. In this large source area emissions have different temporal trends.

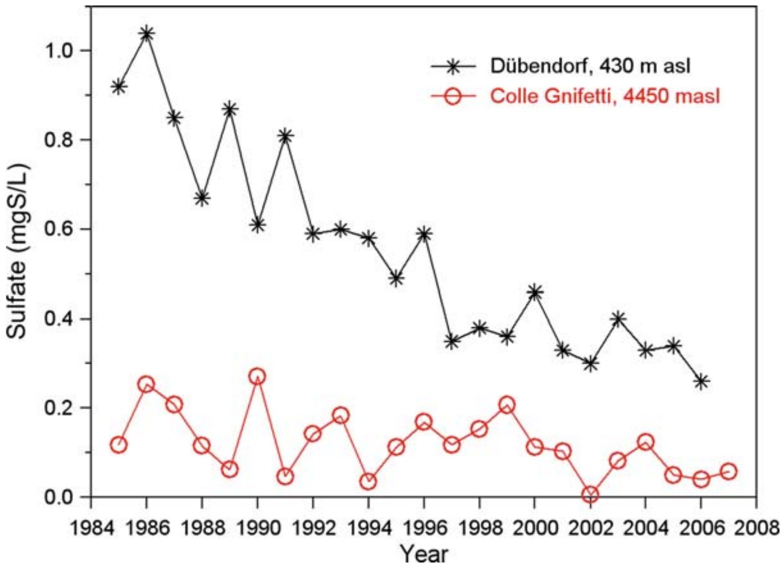

Fig. 6 Annual average sulfate concentrations in snow and ice at the high-Alpine site Colle Gnifetti and in rain at the lower-elevation site Dübendorf. Data of Dübendorf are from Nabel [24]

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Persistent Organic Contaminants in Alpine Waters

John N. Westgate and Frank Wania

Abstract Organic chemicals that are sufficiently persistent are capable of undergoing long-range transport and travel great distances from their sources through the atmosphere. Those organic chemicals that are more efficiently scavenged from the atmosphere in alpine environments than in the surrounding areas can become enriched in mountains, i.e. are found in higher than background concentrations at higher altitudes. This is confirmed by measurements in snow and glacial ice as well as in the waters and aquatic and lotic biota of mountains. Those organic contaminants that exhibit increasing concentrations with elevation are often also those expected to exhibit bioaccumulation and biomagnification, i.e. to achieve higher concentrations at higher trophic levels. As the effects of long-term exposure to many of these chemicals is not yet understood, and as they are – or are structurally related to – known toxins, the presence of higher-than-background concentrations of certain POCs in alpine environments is of considerable concern.

Keywords Alpine lakes, Long-range transport, Mountain cold-trapping, Persistent Organic Contaminants, POPs

Contents

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1 Atmospheric Transport and Mountain Cold-Trapping of Persistent Organic Contaminants

1.1 Persistent Organic Contaminants

Unlike metals and other inorganic species, many organic contaminants have no natural sources – the polycyclic aromatic hydrocarbons (PAHs) being a major exception. As most are also liquids or solids within the environmentally relevant temperature range they were once thought to remain at or near their places of production or use, or, in the case of agricultural chemicals, where they were applied. It is certainly counter-intuitive that concentrations of certain persistent organic contaminants (POCs) would be higher at high elevations, far from their sources, than they are at lower elevations closer to their sources. Yet this phenomenon, sometimes referred to as mountain cold-trapping, has been observed for environmental media as diverse as plant foliage [1, 2] snow [3], and fish from alpine lakes [4].

This is similar to the discovery of high levels of certain POCs in arctic wildlife [5], which eventually led to their ban, or extremely curtailed use, under the Stockholm Convention on Persistent Organic Pollutants (POPs) [6]. POPs are a class of chemicals defined as sharing four properties: they are persistent, i.e. they do not break down readily in the environment; they are toxic to humans and wildlife; they undergo long-range transport (LRT); they bio-accumulate. Indeed, some of the compounds that are subject to mountain cold-trapping are classified as POPs. Here we specifically use *contaminant* to include those chemicals that share the persistence of POPs, but are not currently listed within the Convention, are at concentrations too low to be considered pollutants, may not bio-accumulate and/or whose toxicity is not thoroughly understood.

The POCs include, but are certainly not limited to the polychlorinated biphenyls (PCBs) and the organochlorine pesticides, including those in current use, restricted use and historic use; brominated flame retardants including polybrominated diphenyl ethers; PAHs; and the sometimes more toxic transformation products of these chemicals. Table 1 summarizes information on some of the POCs more commonly detected in alpine environments.

Table 1 Persistent organic contaminants mentioned in the main text

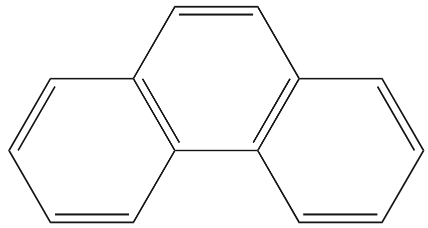
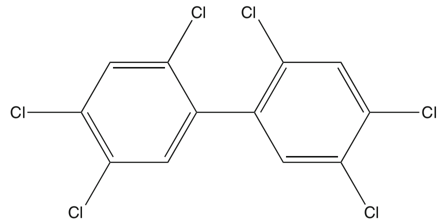
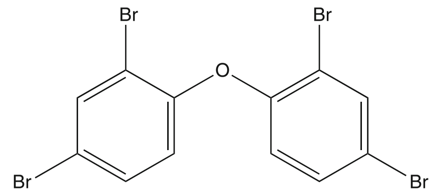
Name	Acronym	Use(s) and/or Source(s)	Chemistry	Example structure[a]
Polycyclic aromatic hydrocarbons a.k.a. polyaromatic hydrocarbon	PAH	Unintentional by-products of combustion of fossil fuels, wood, tobacco and meat, and components of coal tars and other heavy petroleum products. Occur naturally in the environment and are also included in plastics, dyes, pesticides, and medicines	Theoretically unlimited group of chemicals differing in the number and position of the fused aromatic rings	Phenanthrene 
Polychlorinated biphenyls	PCB	Included in industrial insulating liquids and coolants, especially in electrical transformers and capacitors. Also additives in paint, plastics, and carbonless copy paper. Banned under the Stockholm Convention	A suite of 209 possible congeners, differing only in the number or position of chlorine atoms on the two aromatic rings. Often identified by number under a systematic naming convention	PCB 153 (2,2′,4,4′,5,5′-hexachlorobiphenyl) 
Polybrominated diphenyl ethers	PBDE	Included in manufactured plastics and synthetic foams as flame retardants, especially in electronics, carpets, and furniture	A suite of 209 possible congeners, differing only in the number or position of bromine atoms on the two aromatic rings. Often named using the same numerical naming convention as the PCBs	PBDE 47 (2,2′,4,4′-tetrabromodiphenylether) 

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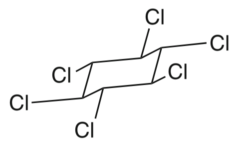
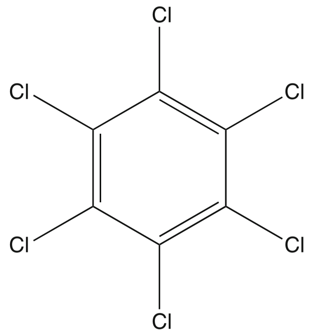
Table 1 (continued)

Name	Acronym	Use(s) and/or Source(s)	Chemistry	Example structure[a]
Hexachlorocyclohexanes	HCH	Insecticides applied to food and forest crops, banned for use in many countries. Technical HCH, a mixture of α-, β-, δ-, ε-, and γ-HCH, is no longer made. γ-HCH (a.k.a. Lindane) is still manufactured and used in shampoos and creams against lice and scabies	Three common isomers, eight possible, differ in whether each of the six chlorine atoms are in the "up" or "down" positions on the nonaromatic ring	*β*-HCH[b] 
Hexachlorobenzene	HCB	Fungicidal seed coating, also produced as a byproduct of other chemicals and incomplete combustion, including automobile exhaust. Banned under the Stockholm convention	There can be only one. The substitution of more electronegative chlorine atoms for the hydrogen atoms of benzene renders that already stable molecule very persistent	HCB 

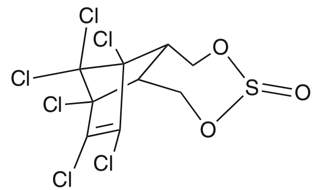
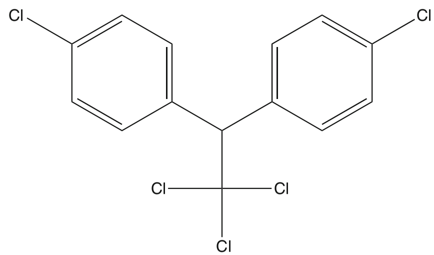
Endosulfans		Insecticide on food and forest crops and a biocidal wood preservative. Still in manufacture and use, especially on tender fruits	Two stereo-isomers and one stable breakdown product, endosulfan sulfate	α-endosulfan (a.k.a. endosulfan-I)[b] 
Dichlorodiphenyltrichloroethanes	DDT	Insecticide, used mostly to protect from diseases spread by insects. Also a byproduct in the production of dicofol, another insecticide. Limited use allowed under the Stockholm convention for areas where malaria is a significant health risk and economic constraints prevent the use of other pesticides	Two common isomers differing in the positions of the single chlorine atoms on each of the two aromatic rings	*p,p′*-DDT[b] 

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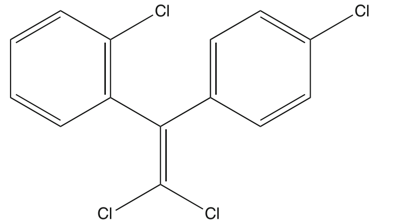
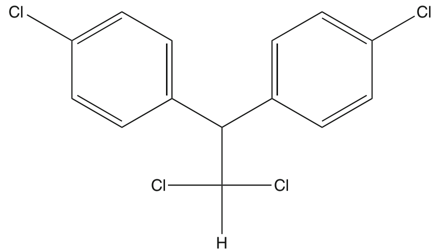
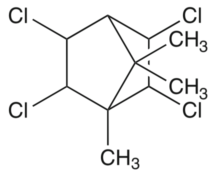
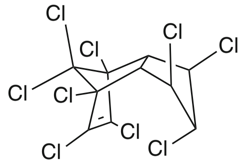
Table 1 (continued)

Name	Acronym	Use(s) and/or Source(s)	Chemistry	Example structure[a]
Dichlorodiphenyldichloroethenes	DDE	A metabolite of DDT, not toxic to insects but likely as mammal-toxic as DDT	Two common isomers differing in the positions of the single chlorine atoms on each of the two aromatic rings	*o,p′*-DDE 
Dichlorodiphenyldichloroethanes	DDD	Insecticide, thought to be less acutely toxic to mammals than DDT. Also used to treat cancer of the adrenal gland. Not specifically banned under the Stockholm convention, but is no longer manufactured in most countries	Two common isomers differing in the positions of the single chlorine atoms on each of the two aromatic rings	*p,p′*-DDD[c] 
Chlorobornanes		The insecticide and piscicide Toxaphene is a mixture of almost 200 chlorinated camphene compounds including many chlorobornanes and chlorobornenes.	Because analytical chemistry techniques separate individual chemicals, or groups of very closely related chemicals, quantifying levels of toxaphene would be	2,3,4,5-tetrachlorobornane[b,d]

	Toxaphene is banned under the Stockholm convention	prohibitively laborious; hence, chlorobornanes are used as a surrogate for measuring relative levels of toxaphene between systems or over time	
Nonachlors	Along with the pure chlordanes, two of 50 compounds found in technical Chlordane, an insecticide used primarily against termites and ants, and on food crops. Chlordane is banned under the Stockholm convention	A pair of stereo-isomers, *cis*- and *trans*-nonachlor can be used as surrogates for measuring relative levels of chlordane between systems or over time	*trans*-nonachlor[b] 

[a]Structure cartoons constructed with ACD/Chemsketch Freeware 10.02; bond-lengths adjusted for easy viewing
[b]Inferred hydrogen atoms not shown
[c]Some inferred hydrogen atoms not shown
[d]Stereo isomerism not shown

1.2 Long-Range Transport and Cold-Trapping

Whereas ocean currents can be involved in organic contaminant transport to polar regions, atmospheric transport is the only plausible source in remote, alpine regions. A clear indication of the importance of atmospheric transport as the source of POCs to remote alpine lakes is the failure to detect in such lakes those organic substances that can not undergo atmospheric transport, but which are prevalent in lakes more directly impacted by run-off and sewage treatment plants. For example Balmer et al. [7] and Schmid et al. [8] did not detect methyl triclosan, a transformation product of the bactericide triclosan, and synthetic musks in fish from remote alpine lakes in Switzerland, respectively. Neither could Balmer et al. [9] detect organic UV filter compounds in the water of a remote Swiss lake. One may even argue that the presence of an organic compound in a remote alpine lake could be taken as evidence that the compound does undergo atmospheric transport.

In both the polar and alpine cases the mechanism of atmospheric transport entails the volatilization or – in the case of species which are gaseous – release of the chemical at its source, transport of the contaminant from its source through the atmosphere, either in the gas phase or sorbed to atmospheric aerosols, and deposition in the polar or alpine environment either directly (dry deposition) or by precipitation scavenging (wet deposition). Figure 1 illustrates in a simplified manner the processes involved in the transfer of POCs along valley to mountain transects during day and night, respectively. Whereas concentrations in the air are expected to decrease (or at worst stay uniform) with increasing distance from

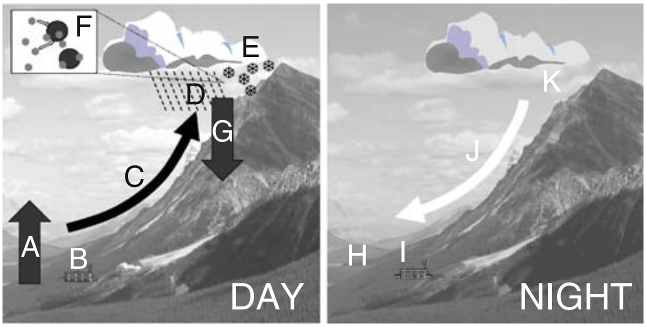


Fig. 1 During the day warmer temperatures increase volatilization of contaminants from surfaces (A) and anthropogenic sources are commonly active (B) in the lowlands. Diurnal heating patterns cause winds to move upslope (C) where water vapor cools and condenses into clouds causing rain (D) and snow (E) which scavenge gases and particles leading to wet deposition. The relatively colder mountain air enhances sorption of contaminants to aerosols (F) augmenting dry particle-bound deposition and the cool highland surfaces have a greater capacity for contaminant retention which increases dry gaseous deposition (G). The sum is a net movement of contaminant to high mountain surfaces driven by a strong vertical temperature gradient. At night there is little energy for the volatilization of contaminants from surfaces (H) and many anthropogenic sources are inactive (I). Cooling causes surface winds to move downslope (J) which keeps water vapor below the saturation vapor pressure and dissipates clouds (K). Thus, the weak vertical temperature gradient causes little net movement of contaminants between low and high areas

source, the concentrations in surface phases depend on the relative rates of atmospheric deposition and loss from the surface. Only when some factor causes preferential deposition and/or retention of a chemical in an environment will that environment show enrichment of that chemical that is greater than the background level of contamination [10, 11].

An obvious correlation between polar and alpine environments is the decrease in temperature with increasing latitude or elevation. This temperature change leads to a shift in environmental phase distribution equilibria – i.e. a chemical moves from the atmosphere to terrestrial surfaces, including direct deposition to surface waters, but also to snowpack and soils from which movement into surface and groundwater is possible. This process has been termed "cold condensation" but should more correctly be called "cold-trapping" because the contaminants are not actually condensing.

Despite some apparent parallels, different mechanisms are believed to be responsible for cold-trapping in polar regions and in mountains [12]. The temperature dependence of the phase distribution between atmosphere and the Earth's surface, in conjunction with the latitudinal temperature gradient and large-scale atmospheric circulation, is believed responsible for the cold-trapping of POCs at the poles [13]. Mountain cold-trapping on the other hand, is believed to be caused by more efficient precipitation scavenging of atmospheric POCs at higher than at lower altitudes [12]. This can simply be brought about by the altitudinal temperature gradient, as the temperature dependence of the air–water distribution equilibrium implies that cold rain is more efficient in scavenging many POCs than warm rain. Cold-trapping in cold mountains will also occur, if a POC is more efficiently scavenged from the atmosphere by snow than by rain. Finally, POCs can be more efficiently scavenged at higher altitude if they become associated with particles at the lower atmospheric temperatures prevailing at higher altitude. This is because rain and snow are efficient in scavenging particles from the atmosphere. Obviously, mountain cold-trapping is expected to be particularly pronounced when the rate of precipitation increases with altitude, which is often the case in temperate mountains. Not all POCs will be subject to mountain cold-trapping, but only those whose rate of wet deposition is sensitive to the changes in temperature that occur along a mountain slope [12]. Model simulations and field data suggest that POCs subject to mountain cold-trapping are considerably less volatile than POCs that get enriched at high latitudes [12].

2 Delivery of Persistent Organic Contaminants to Alpine Waters: Atmospheric Transport and Deposition

2.1 Air Sampling Methods

In order to understand how POCs travel from their sources to the waters of alpine environments, several studies have investigated air concentrations of POCs either at one high altitude location as a function of time or at different elevations of a mountain, providing vital clues to their behavior. The concentrations of POCs in the alpine atmosphere are usually low. Unlike condensed phases (water, snow, and sediments), the volume of air samples needed for a reliable quantification can therefore not realistically be transported from the field to the laboratory. Instead of direct sampling, the sorption of contaminants to some solid phase is employed. The air-sampling techniques commonly used for organic contaminants can be divided into two main categories: active high volume sampling and passive air sampling.

High volume air sampling is achieved by drawing air through sampling media using a vacuum pump [14], and thus requires a source of electricity to operate. This method is able to sample a large volume of air in a relatively short time, so that it takes a "snapshot" of the air concentrations. The sampling media usually consist of a glass fiber filter (GFF), which traps particles, followed by an open-celled foam disk or plug, usually poly-urethane foam (PUF), which sorbs organics from the vapor phase. In some experiments the PUF may be coated or complemented by XAD-2 resin, a copolymer of long-chained polystyrenes and divinylbenzene which has a higher capacity for POCs than PUF alone. After sampling, the media are solvent extracted and the extracts concentrated for analysis. If the GFF and disks are extracted and analyzed separately it can be determined which POCs are associated with particles and which are in the gas phase in the atmosphere.

Passive air samplers (PASs) for POCs rely on diffusion to move contaminants to the sampling medium, and thus do not require a source of electricity, making them far more portable, less expensive, and easier to operate. However, in order to sample a large enough volume of air, diffusive PASs must be deployed for weeks to months at a time. A PAS thus provides time-averaged air concentrations over fairly lengthy time periods. The sampling media for PAS do not include a GFF and few, if any, particles are sampled. The diffusive PASs use either a PUF plug [15], an XAD-2 resin [16], or a semipermeable membrane device (SPMD) in which a film of triolein – a neutral synthetic lipid – is placed within a polyethylene membrane tube [17]. Diffusive PUF samplers typically are deployed for a maximum of 4 months, XAD-2 around 12 months, and SPMDs for 24 months

The independence from a power supply makes PASs popular options for measurements of POCs in the remote alpine atmosphere, but they also limit the temporal resolution that can be achieved. Trying to combine the advantages of both PAS and active samplers, a flow-through sampler (FTS) was recently designed, which requires no external power source, but can sample large volumes of air in fairly short periods of time [18, 19] by rotating into the wind and have it blow through a series of PUF disks. The volume of air sampled can be estimated from wind speed records. The FTS will trap particles, but they cannot be analyzed separately from the gas phase POCs.

2.2 *Studies Measuring POCs in Mountain Air*

A number of studies have taken high volume air samples at sites located in mountains, generally with the intention of identifying source regions of the contaminants being transported to an alpine area [20–27]. The field study by Primbs et al. [26, 27], based on Mt. Bachelor in the Cascade range of Oregon in the American Northwest, illustrates the information that can be derived from a large number of high volume air samples taken on a mountain. Careful analysis of the short-term temporal variability, in the POC concentrations in conjunction with air mass trajectories allowed the identification of distant and regional source regions. Specifically, elevated concentrations of hexachlorobenzene (HCB), α-hexachlorocyclohexane (α-HCH) and particle-bound PAHs could be linked to trans-Pacific transport from Asia, whereas urban areas in California are fingered as contributing significantly to the presence of gas-phase PAHs and fluorotelomer alcohols at the sampling sites. Elevated concentrations of PCBs and a number of pesticides (γ-HCH, chlordanes, α-HCH, HCB, and trifluralin) were associated with fires in Western North America due to revolatilization of these POCs from soils and vegetation [26, 27].

Longer term air concentration variability, such as occurs on a seasonal scale, can also yield insight into the sources of POCs, although for pesticides such variability is often dominated by their seasonal application, confounding an elucidation of transport mechanisms [23]. For some out-of-use chemicals a seasonal dependence of air concentration has been observed that appears to stem from the relative stability of summer and winter air masses; this seasonal difference delivers more "continental" air to the mountains during the warm months, and more "oceanic" air during cooler months. This is supported by the winter months showing a proportional increase in α-HCH over the other HCH isomers; α-HCH has such high oceanic concentrations that oceans can be considered a source [23].

While clearly superior when seeking high temporal resolution, high volume samplers are less suitable to observe spatial differences within mountains and along altitudinal transects. Nevertheless, with sufficient logistic effort it can be done. In the United States, for instance, air concentrations of organic agricultural pesticides were highest in the Californian Central Valley, and lower at several different elevations in the Sierra Nevada Mountains [20]. This pattern of high air concentration near the source, falling to a low – but not zero – asymptotic background concentration at sufficient distance is typical for those POCs that are still in use. Most studies quantifying POC air concentrations along elevation gradients have relied on PASs [28–34]. For those substances that have a source at the base of the mountains, these PAS studies also observed concentrations that decrease with elevation. For example, air concentrations of PAHs and the pesticide Lindane, which is nearly pure γ-HCH, decreased from Costa Rica's central valley to higher elevation sites on the surrounding volcanoes [30, 31]. Similarly, PAH air concentrations decreased in the mountains of Western Canada with increasing elevation and therefore increasing distance from the sources associated with vehicle traffic and small human settlements [34].

Relatively uniform concentrations of POCs along an altitudinal transect suggest the absence of significant local sources. For example, concentrations of pesticides in the air of mountains in Western Canada [32], which are further from the source regions than are the Sierra Nevada Mountains from the Californian Central Valley, have a weak or absent relation to elevation, hinting at LRT as the dominant source. The same was observed for HCB and α-HCH in the Cordillera Central of Costa Rica [30].

Some measurements suggest that POCs may even occur at higher concentrations in colder air higher in the mountains [28, 29, 32]. Studies on mountains in relatively close proximity found that air concentrations of α-endosulfan, *p,p′*-dichlorodiphenyldichloroethene (DDE), a metabolite of dichlorodiphenyltrichloroethane (DDT), and chlorinated benzenes measured by PAS exhibit this behavior even under a variety of precipitation regimes. The reason for upslope increases in gas phase air concentrations is unclear and may at least partially be attributed to sampling artefacts [32]. However, for those POCs that no longer have primary sources or those that are measured at a sufficient distance from primary sources it is conceivable that the surface media may act as a secondary source, allowing chemicals to revolatilize when temperatures are sufficiently high. If the rate of revolatilization is higher at higher altitude, this could explain increasing air concentration trends with elevation.

2.3 Measurements in Snow and Glaciers

While air concentration measurements can confirm the possibility of atmospheric transport to alpine regions, POCs have to be deposited to the surface to reach alpine waters. This can occur by a variety of routes, including wet and dry deposition of both gaseous and particle-bound POCs. Some studies have sought to directly measure atmospheric deposition [35, 36] despite the considerable logistical challenges. Because snow is the major form of precipitation and snow and ice constitute the receiving surface in many alpine regions for most of the year, POC deposition in mountains is often quantified by measuring concentrations in snow and ice [3, 37, 38]. In particular, by sampling the spring time snow pack, it may be possible to capture the net atmospheric deposition that has occurred over the preceding winter without incurring the cost and logistical effort of deploying sampling equipment at high altitude.

However, many factors may influence the concentrations of POCs in snow and ice. Samples taken prior to first melt in temperate mountains have shown higher concentrations of POCs than the same snow sampled after some melting, suggesting a rapid release of POCs from snow upon melting [39]. Furthermore, snow sampled from forested areas can show significantly lower concentrations of POCs compared to snow from open or sparsely vegetated areas such as glaciers [39]. Additionally, year-round bulk deposition sampling in alpine areas, which collects snow, rain, and aerosols [35, 39], shows some seasonality, with increased deposition of POCs in warmer periods, suggesting that summer time deposition can be significant at higher altitudes. Seasonal application of current-use POCs could explain higher summer time deposition for some analytes, but not for past-use POCs, which are assumed to revolatilize in warm weather and become once again available for atmospheric deposition [39].

Measurements of POCs in the snow and glacial ice of mountains have revealed that anthropogenic contaminants are not only present in these media, but that some occur at higher concentrations at higher elevations than at lower elevations [3, 40]. Specifically, the HCHs, lower chlorinated PCBs and endosulfan showed this trend in the mountains of Western Canada [3], while p, p'-DDT and p, p'-DDD did so in snow collected from four elevations on Mt. Everest [40].

Hageman et al. [41] correlated pesticide concentrations in mountain snow from various Western US National Parks with the amount of farmland within a given distance of a sampling site to determine the source of the pesticides. A strong correlation between the two appears for current-use pesticides, showing the importance of local use areas, but a weak correlation applies for historic-use pesticides, indicating that LRT is important.

Measurements of POCs concentrations in ice cores retrieved from alpine glaciers promise to provide information on the temporal trend in the atmospheric deposition, both on a seasonal [42] and a decadal time scale [43, 44]. Extensive studies were undertaken on the high altitude, high accumulation Lys glacier on Monte Rosa in the European Alps [42, 43] and on a number of high-altitude glaciers in the Himalayas [40, 44, 45]. Analysis of the surface layers of the Lys glacier at relatively high resolution did not reveal seasonal trends for most POCs. Only lindane, which was still in use in Europe at the time had higher deposition rates during summer [42]. On the other hand, seasonal variations in the deposition of PAHs were noted in a Himalayan glacier [45]. The concentration trends for DDT, HCHs, and PAHs observed for the latter half of the twentieth century in a glacier on Mt. Everest were largely consistent with what is known about the changes in the source strength of these POCs in India [44], lending credibility to the usefulness of glaciers as historical archives.

3 Transport and Distribution of Persistent Organic Contaminants in Alpine Waters

Because of the very low concentrations, the detection and quantification of POCs in the water of alpine lakes and streams is very challenging [38] and is therefore rarely attempted. Nevertheless, studies of POCs in lakes in Europe show HCH concentrations in water that rival those of the highest measured in inland waters [46]. Mountain lakes close to populated areas showed PAH profiles which favored the lighter PAHs, including some transformation products, whereas a more remote lake contained relatively higher concentrations of heavier PAHs, a hallmark of LRT because the lighter PAHs are generally the more labile [47]. The concentration of POCs in water and sediments of alpine lakes is dynamically controlled by the rates of input and loss. In order to understand the contamination of the alpine aquatic biota, it is important to understand the processes that deliver POCs to the lakes and that lead to their loss from the lakes (Fig. 2). Two alpine lakes in particular have

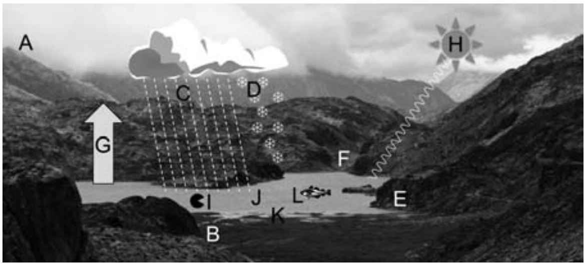


Fig. 2 Water, and thus contaminants, enter alpine lakes in various ways including melt water from glaciers (A), alpine streams (B), precipitation directly onto the surface (C and D), and from groundwater (E). Routes out of the lake include output streams (F), groundwater and volatilization (G). Transformative losses include photo-(H) and bio-(I) degradation. Storage in the lake can occur in the water column (J), sediment (K), or biota (L)

been characterized in terms of their mass budget for POCs – Bow Lake in the Canadian Rocky Mountains [39, 48, 49] and Lake Redo in the Pyrenees [50].

Only a fraction of the atmospherically derived POCs is directly deposited onto alpine lakes. Other inputs occur with run-off from the seasonal snowpack and from melting glaciers, as well as from rain. The flows into an alpine lake can be classified as of either glacial or direct precipitation/groundwater origin. Measurements of both types of stream flowing into Bow Lake in Alberta, Canada have shown that the primary source of PCBs, HCHs, and the pesticides endosulfan and dieldrin to Bow Lake is glacial runoff [49]. The other major input route is direct gas absorption by the water. Levels of HCHs in the glacial runoff were on the order of four times that of streams nearby. The concentrations of the other POCs were higher in the runoff too, but less so. Blais et al. [48] estimated that glacial runoff was responsible for 50–97% of the organochlorine inputs to Bow Lake, compared with 73% of input water. Similar differences between glacial and nonglacial streams have been observed in the European Alps [51]. Interestingly, HCB levels were higher in one of the precipitation/groundwater streams into Bow Lake than the glacial runoff [49]. This highly volatile POC is found in similar concentrations in air the world over, and may volatilize from the snowpack before melt occurs, leading to lower concentrations. Further support for glacial melt as a major source of POCs in alpine lakes comes from a lack of correlation between annual snow pack burden and stream concentrations [39]. The extent of glacial melt is likely a more important determinant of POCs input during a particular year [39].

The importance of melt-water to the POC load in alpine lakes is a matter of some debate. It has been argued that direct deposition and the melting of snow that falls onto the frozen surface of a lake are the chief sources of POC transport to a lake [50]. The grounds are that movement by groundwater into the lake is unlikely as POCs likely sorb to soil organic matter, and that, at least for the lake studied, most of the water input by streams does not mix with the rest of the water column before flowing out again. The transport of soil into a lake by erosion is likely not an important process under normal conditions and this is confirmed by differences in radionuclide measurements of soils around and sediments in the lake [52]. However, the ability of soils to trap POCs is highly variable, and not well understood under melting conditions. It is possible that ice dams within a melting snowpack and frozen soil prevent melt-water movement into the soil, and instead promote direct movement to lakes and streams. Unfrozen soils may release old contaminants and exchange them for new as melt-water passes through. Finally, it is unclear how POCs directly deposited from the atmosphere would cross the swifter moving epilimnion to mix with the water column any more efficiently than POCs entering with a stream.

There are several processes by which POCs are lost from alpine lakes. Besides transformation into other chemicals, POCs in alpine waters may volatilize from the surface of the lake, may flow out through a stream, or may become trapped in sediments. In Bow lake volatilization, outflow, and sedimentation were all found to be important [49]. The transfer across the air–water boundary is a dynamic one, the direction of which is controlled by the relative concentrations of each POC in the two bulk phases: the system will tend towards equilibrium, but the rate of transfer between the two phases can be slow enough that it may never be established. An interesting case is found in Bow Lake, where the flux of γ-HCH, among other POCs, was measured over two years [53]. At the beginning of the study the air concentration of γ-HCH was high because of spring application of this pesticide by farmers, so the net movement of γ-HCH was from air to the water. Over the next few months, as air concentration dropped and temperature rose, equilibrium was reached between the two – i.e. the net flux of γ-HCH between air and water was zero. The following spring saw a large increase in melting from Bow Glacier, which feeds the lake, and this increased melt released a substantial amount of γ-HCH into the lake, at which point the net flux of γ-HCH across the air–water interface was out of the water [53].

Studies of the levels of POCs downstream of alpine lakes are few, confounding an understanding of the contribution of outflow to the lowering of POC levels in those lakes. The Laja watershed of Southern Chile consists of a glaciated mountain, the melt of which feeds Lake Laja, which is at the head of the Laja River [54]. The level of anthropogenic activity – forestry, agriculture, and industry – steadily increases downstream. Measurements of POCs in those waters found none of the pesticides for which analysis was made, but found levels of PCBs in the lake that were twice as high as levels measured in industrialized lakes in North America. However, PCB levels in the outflow stream were below detection limits. It is possible that the PCB load in the lake is caused partly by a hydroelectric plant at that site as well as LRT.

In the Bow River basin, estimates of total fluxes of some POCs based on measured concentrations in inflow, outflow, and air suggest that for most POCs volatilization is more important than outflow in removing contaminants from the lake [49]. However, mass balances suggest that outputs are somehow overestimated by the methods used, and it is unclear in which loss process or processes this overestimation lies or lie. Nonetheless, the measured concentrations of all POCs analyzed were lower in the outflow water than in the glacial inflow, but higher than the other inflow streams.

4 Implications of the Presence of Persistent Organic Contaminants in Alpine Waters

4.1 POCs in Alpine Aquatic Organisms

Already in 1980, Zell and Ballschmiter [55] reported on the presence chlorobornanes – which make up the mixture known as toxaphene, a POP – in Arctic char (*Salvelinus alpinus*) from an alpine lake in the Tyrolian Alps. However, concern about the presence of POCs in fish from alpine lakes would not be raised until PCBs, toxaphene and other pesticides were measured in water, sediment, and fish of several mountain lakes in Western Canada much later [56, 57]. Five of the studied lakes had been intentionally treated with toxaphene in the late 1950s and early 1960s to remove "undesirable" fish so they could be stocked for sport fishing, but all of the lakes studied contained some chlorobornanes. Concentrations of chlorobornanes in the water and sediments of most untreated lakes were below the analytical detection limits, but the concentrations in the fish of those lakes were much higher [57]. The concentrations of chlorobornanes in fish had a significant negative correlation with measures of lake productivity. Intriguingly, the highest concentrations of POCs found were in fish from Bow Lake, an untreated, glacially fed lake. That same lake was also the only untreated lake that had quantifiable levels of chlorobornanes in water and sediment [57]. The counterintuitive observations of POC concentrations in fish from alpine lakes that increase with altitude [4, 58] remain the most compelling illustration of the mountain cold-trapping phenomenon. In particular, the studies on contaminants in fish from a variety of European mountain lakes provide overwhelming evidence of that phenomenon [4, 59–64].

The easy detection of POC in alpine fish is the result of two processes that occur if a chemical partitions readily into lipids from water and is not efficiently metabolized. Some POCs will bioaccumulate, meaning that their concentrations in organisms will be much higher than in the surrounding water, and some will biomagnify, meaning that their concentrations in organisms will increase from one trophic level to the next [65]. Some organic contaminants such as the PAHs do not readily bioaccumulate because they are subject to fairly rapid metabolization. They have nevertheless been detected in aquatic organisms and the livers of fish from alpine lakes [66, 67]. The composition and concentrations of PAHs in the biological samples is then often controlled by the metabolic capability of a particular organism [67].

Grimalt and coworkers have used a variety of data interpretation techniques to show the influence of lake temperature and lake altitude on POC concentrations in fish [4, 59, 60, 63, 64]. It is important to bear in mind that all sorts of factors may be contributing to differences in the concentrations of POC in fish from different lakes, including the distance of the lakes from POC source regions [11]. Such factors may confound a interpretation of increasing POC concentrations in fish with altitude in terms of mountain cold-trapping. For example, the age of organisms is a factor in the contaminant load [68]: older organisms are more contaminated than young ones, at least where steady state between organism and surrounding is not reached.

Differences between the amount of biomagnification between two lakes can sometimes be explained by different lengths of food chains [69] – i.e. the top predator in a lake with three trophic levels will have a lower contaminant load than the top predator in a lake with five trophic levels, even though they are of the same species. An analysis of the food web of Bow Lake revealed that it had only three trophic levels so food chain length could not explain the high POC loads of fish there [70]. It was found that fish that relied on pelagic (open water) sources for food had higher loads than fish of the same species that relied on benthic (near and on the bottom sediments, including near shore) sources because of the higher lipid content, and thus the higher POC load of the food source. As lipid content of food animals correlates negatively with lake temperature, and thus positively with elevation, this effect is magnified in alpine lakes [71].

Biodilution is the decrease in concentration of contaminant caused by an increase in organism mass; faster growing organisms will have less contaminant per unit mass or volume than slower growing organisms that have the same uptake rate of chemical, although the total load in both organisms will be the same. Because the growth rate of organisms is slower in colder lakes, the effects of biodilution are smaller in higher lakes [71], adding another mechanism by which POC loads become elevated in alpine aquatic organisms. Indeed, Blais et al. [71] observed that "the effect of elevation, lipid content, and temperature on contaminant concentrations was no longer significant once the data had been adjusted for variable growth rates" of amphipods sampled from lakes at different altitude.

The exact effects of current levels of POCs on the health of fish are not entirely understood, but it has been shown that fish in alpine lakes with greater POC concentrations exhibit greater signs of oxidative stress than fish in lakes at lower altitudes [58]. When investigating the estrogenic activity of fish muscle extracts from ten European mountain lakes, Garcia-Reyero et al. [72] observed a correlation with fish age and concentrations of the heavy PCBs. Specifically, fish with high estrogenic activity were found in lakes containing high levels of POCs in the sediments. This is consistent with a strong correlation between estrogenic activity and the presence of POCs, including heavy PAHs, that were observed in a study of sediments from 83 European mountain lakes [73].

While fish may be the top of the underwater food web, they are sometimes themselves consumed by aquatic birds. Indeed, it was in aquatic birds that the dire effects of POPs on wildlife were first noticed [74]. The birds that rely on alpine waters for their food are similarly contaminated with POCs. PCBs, DDE, HCB, and the OC insecticide *trans*-nonachlor have been shown to biomagnify and are found in the eggs of American dippers (*Cinclus mexicanus*), a lotic (flowing water) bird [75]. Osprey eggs from alpine catchments in Western Canada contained a variety of POCs at higher concentrations than in fish from the same sites but below levels of concern for reproduction [76]. Lower POC concentration in prey in the osprey's wintering grounds may dilute the contaminants taken up with alpine fish in summer.

4.2 Concerns for Humans

As early as 1961 scientists looked for and found measurable levels of POCs, indeed POCs that would become classified as POPs, in human fat and by 1965 in human breast milk [77]. While something of a surprise at the time, the current understanding of bioaccumulation and biomagnification explain this, as humans tend to eat high in the food web. For people reliant on alpine sources for food then, it is possible that they are exposed to higher levels of certain POCs, i.e. those that are subject to efficient mountain cold-trapping, than are people nearer to the POC sources.

The increased load of POCs in fish from alpine lakes is of obvious concern for those dwelling in alpine environments that use these fish as food, but also for sport fishers on alpine lakes. In the case of the mountains of Western Canada, it is ironic that the use of toxaphene to improve sport fishing has rendered those fish less safe to consume even decades later. Cattle and domesticated yaks and goats are also raised in alpine environments, both for meat and for dairy products. In the European Alps, cattle are often moved to higher grazing grounds in the summer months. Mountain cold-trapping may result in higher concentrations in the summer feed of those cows, and the human consumption of these animals and their products is a potential route for increased POC exposure. A study of the influence of grazing altitude on the POC levels in the milk of dairy cattle in the Swiss Alps is ongoing [78].

Controlled toxicological studies of the effects of POCs on humans cannot ethically be carried out. It is also likely that the combined effects of several POCs will differ from a simple sum of the effects of individual chemicals, so the understanding of the threat of POC exposure to human well-being is incomplete. So-called cohort studies, in which the health of a group of people are followed over time, have shown correlations between maternal POC levels and decreases in gonadal hormone levels in newborns [79]. Cohort studies of adults indicate that higher POC levels are associated with decreased gonadal hormone levels as well [80]. However, any evidence from such studies is necessarily circumstantial as other environmental and genetic factors cannot be controlled for. The eventual effects on long-term reproductive health have yet to be elucidated.

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Part III
Ecology of Alpine Waters

Water Sources and Habitat of Alpine Streams

A.M. Milner, J.E. Brittain, L.E. Brown, and D.M. Hannah

Abstract We review the different water sources of alpine streams, namely snowmelt, glacial meltwater, and groundwater. Alpine water sources have different physicochemical properties, with their relative contributions determining habitat characteristics of receiving streams. High spatio-temporal variability of water source contributions makes rivers in alpine glacierized basins distinct from other lotic systems. The timing and volume of bulk (snow and ice) meltwater production, along with inputs of groundwater generate distinct patterns of stream discharge, water temperature, suspended sediment, hydrochemistry and channel stability over annual, seasonal and diurnal time-scales. These temporal changes in water sources are sensitive to anthropogenic pressures including climate change, water resource allocations, and contaminants which will ultimately influence water quality and habitat suitability for biotic communities. Heterogeneity of the physicochemical environment creates a spatial and temporal mosaic of stream habitats related

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to differences in water source contributions, and hydrological connectivity. Hydrochemical characteristics are strongly influenced by seasonal snowmelt, and the development of glacier drainage systems. River ice is a particular feature of alpine streams, creating unique environmental conditions that strongly affect the flora and fauna, both directly, and indirectly through changes in their environment. The extent, nature and duration of river ice varies widely across alpine areas. We outline the characteristic longitudinal patterns in benthic macroinvertebrate communities downstream from these water sources created by this physical template.

Keywords Alpine, Biota, Habitat, Hydrochemistry, River ice, Water sources

Contents

1 Introduction

Alpine river systems are fed by snowmelt, glacial ice melt and groundwater [1], with intermittent inputs from rainfall events during late spring to early autumn. These water sources individually and jointly impart characteristic discharge regimes and a distinctive suite of physicochemical properties on receiving streams [2]. Stream physicochemical properties (including discharge/ flow, water temperature, suspended sediment concentration, hydrochemistry and channel stability) can change over temporal scales ranging from diurnal to inter-annual [3, 4]; and they are a major determinant of biotic community structure. This chapter outlines firstly some of the key characteristics of alpine water sources and discusses their influence on diurnal and seasonal changes in stream physicochemical habitat. Thereafter, the chapter considers recent attempts to integrate water source dynamics and physicochemical habitat variability in a holistic framework.

2 Alpine Stream Water Sources

Variability in the magnitude, timing and duration of snowmelt, glacial icemelt and groundwater contributions plays an important role influencing alpine basin hydrology [5, 6]. Glacial hydrology has been studied in detail since the early 1980s [7] whereas, more recently, the role of water source dynamics in a wider alpine basin context has received attention [4, 6]. Contributions of snowmelt, glacial icemelt and groundwater to streams are interlinked when considered in a holistic basin-wide hydrological framework. Meltwaters and groundwater interact over time and space to produce a dynamic riverscape of contrasting hydrological, geomorphological and physicochemical conditions [1, 8]. This section summarises current understanding of each water source in turn as a context for subsequent sections that outline the physicochemical characteristics of streams, and ongoing attempts to create a holistic conceptual framework for linking water source and physicochemical dynamics.

2.1 Snowmelt

Alpine river basin hydrology is dominated by the influence of snowmelt, although the magnitude of stream flow contributions are seasonally variable. In winter, snowpacks cover almost the entire alpine zone (i.e. area above the treeline). During the spring, snowmelt exerts its greatest influence on stream flow. In the ablation zone, snowpacks retain meltwaters during the early summer melt season, thus moderating peak stream discharge [9]. Tributary valley streams also flow during spring – sourced by snowpacks; but some of these streams can be ephemeral and, as summer progresses and snowpacks deplete, they cease to flow. As the transient snowline recedes upvalley throughout summer, glacier ice is uncovered and supraglacial meltwater channels begin to develop on the ice surface. In late summer, snowpacks are typically confined to glacier accumulation zones, highest peaks or north-facing hollows, although occasional summer snowfalls may deliver new snow at lower altitudes. During late summer and early autumn, snowpacks contribute relatively little to bulk stream flow and where glaciers are present; ice melt provides the greatest proportion of meltwater.

2.2 Glacial Meltwater

Over 100,000 glaciers are known from throughout the World; many of these lie in alpine environments [10]. The surface of a glacier consists of an ablation zone comprising ice and seasonal snowpacks at lower elevations, and snow (above the firn line) in the accumulation zone at higher elevations. Large-scale atmospheric circulation influences glaciers by supplying and removing mass (i.e. precipitation) and energy at the glacier and snowpack surface [11]. Different air masses, with contrasting thermodynamic properties, significantly affect snow- and ice-melt. Typically, cyclonic weather patterns are associated with low ablation, whereas high melt is associated with anticyclonic systems [12]. At smaller spatial scales, shading, altitude and transient snowline position (surface albedo) influence the distribution of energy receipt and, in turn, ablation patterns across glaciers [13, 14].

Peak glacier melt contribution to stream flow in the northern hemisphere is in mid- to late-summer when seasonal snow packs have receded to higher altitude. Exposure of ice allows fast drainage of water, as well as meltwater from snowpacks located in the accumulation zone [11, 13]. Some meltwater drains over the glacier surface directly into proglacial streams but a large proportion enters the glacier interior (englacial) drainage system through crevasses and moulins. Water reaching the glacier bed enters the subglacial drainage system and may be routed to the glacier snout in major arterial conduits, smaller conduits linking cavities and/ or thin water films [7, 15]. As the englacial and subglacial drainage systems expand and become more connected over the melt season, diurnal fluctuations in proglacial stream discharge become more pronounced [11, 16, 17]. As water moves through the glacier drainage system, it is physicochemically altered due to entrainment of fine sediment and geochemical weathering [18, 19].

2.3 Groundwater

Groundwater systems in alpine systems can be classified broadly as: (1) alluvial – those areas where water is contained in subterranean sediments along the valley floor, (2) hillslope – valley sides where thin soils and scree deposits store water, and; (3) karstic – underground cave networks formed by the dissolution of carbonate rocks. With distance from the glacier margin, meltwater influences typically decrease and snow- and ice-melt runoff mixes with groundwater (typically from alluvial and hillslope sources) to change alpine stream hydrology [5, 6, 20]. Traditionally, alpine basins have been regarded as predominantly surface water fed (i.e. snowpacks and glaciers) but a recent study of water quantities from different sources in the Taillon-Gabiétous basin, French Pyrénées, revealed that groundwater contributed ~25% of stream flow only 1.5 km from the glacier snout [6]. Groundwater contributions to alpine stream flow are buffered and do not exhibit strong annual variation, with peak contributions lagging meltwater contributions due to recharge and subsequent discharge of aquifers by meltwater [5, 6]. During summer when precipitation falls mainly as rain, groundwater systems can be recharged but stream discharge typically returns fairly quickly to baseflow. Although groundwater sources are strongly dependent on meltwater and precipitation recharge [21], water becomes physicochemically modified due to the long residence times of water in subsurface aquifers [6]. During winter, groundwater inputs dominate stream flow across alpine basins [5, 8].

2.4 Episodic Flows

Episodically, high magnitude inputs such as rainfall and internal storage outbursts may dominate stream flow. Rainfall events are common in alpine river systems, punctuating the characteristic diurnal and seasonal flow pulses that are otherwise characteristic of these systems [4, 17]. For example, Brown & Hannah [22] identified 21 rainfall events over a 140-day monitoring period (i.e. 70 days in both the 2002 and 2003 summer melt seasons) that were at or above the 75th percentile of all rainfall events. The majority of these events had little effect on stream flow, although six lead to 5-fold or greater increases in stream discharge. However, due to combinations of steep valley slopes, large expanses of bed rock/scree and thin soils, runoff from these events was rapid and diurnal stream flow fluctuations driven by meltwater production were restored within 24 h for even the most extreme rainfall events. Major rainfall-induced floods may result later in the ablation season when the winter snow cover and its retention capacity are at a minimum and, at the same time, the glacier drainage system is well developed [23]. Alternatively, summer snowfall may retard ablation and cause a reduction in discharge lasting several days [9]. Glacier outburst flood characteristics and mechanisms are reviewed by Tweed and Russell [24]. Random shorter time scale discharge events indicative of instability within the glacier drainage system have also been recorded (e.g. [25]).

3 Physiochemical Properties of Alpine Streams

Alpine water sources have different physicochemical properties with their relative contributions determining habitat characteristics of receiving streams. High spatio-temporal variability of water source contributions makes rivers in alpine glacierized basins distinct from other lotic systems [4]. The timing and volume of bulk (snow and ice) meltwater production [26], along with inputs of groundwater from springs, seeps, upwellings [27] and hillslopes generate distinct patterns of stream discharge, water temperature, suspended sediment, hydrochemistry and channel stability over annual, seasonal and diurnal time-scales [3, 28]. Heterogeneity of the physicochemical environment creates a spatial and temporal mosaic of stream habitats related to differences in water source contributions, and hydrological connectivity [5, 8, 29]. The spatio-temporal dynamics of these major physicochemical variables are outlined below.

3.1 Stream Discharge

Many alpine rivers have a characteristic annual flow regime with peak flow in spring-summer. However, this “classic” flow regime can be punctuated by episodic

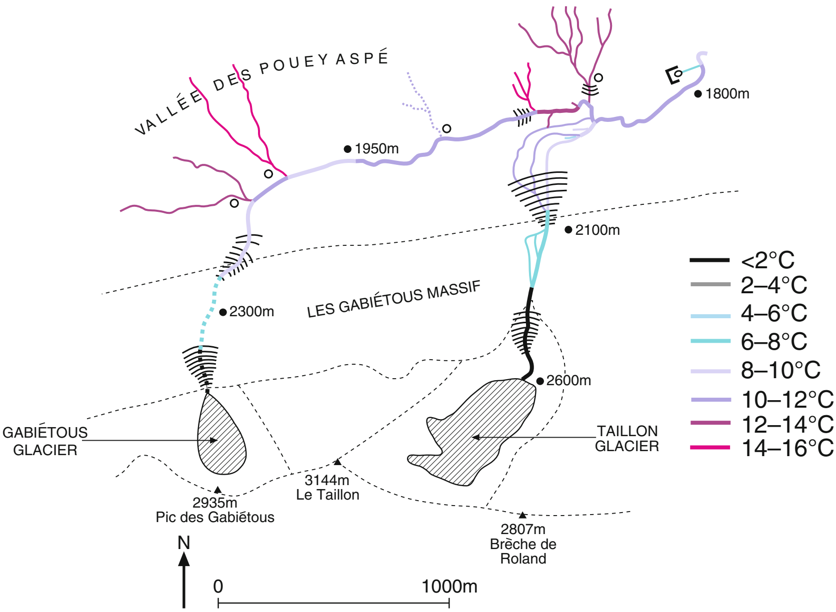

Fig. 1 Spatial distribution of mean water temperatures in the Taillon catchment, French Pyrenees [33]

precipitation events to yield a more variable discharge regime [4, 11, 17]. The seasonal discharge pattern is due to the compensating effect of winter precipitation storage in the solid phase followed by release as summer melt [30]. During the ablation season, runoff exhibits repeated cyclic fluctuations at the daily time-scale in combined response to snow/ice surface energy balance and drainage system processes [11, 17]. Changes in glacierized alpine basin drainage processes can be inferred by classifying diurnal outflow hydrographs based on their form (shape) and magnitude (size) [11, 16, 17]. During the early melt season when snowpacks cover large areas of alpine basins, diurnal hydrographs are characterised by flows that build over the day, peaking in the late evening as a result of snowpacks delaying meltwater movement to the proglacial stream. During the later melt season (July–August) mid-afternoon peak flows (linked to energy receipt at the glacier surface; [13]) are characteristic of glacier-fed rivers (Fig. 1). In contrast, many alpine groundwater streams are characterised by relatively constant discharge over inter-annual time-scales [27], although Brown et al. [31] suggest that some alpine groundwater streams can freeze for short periods during winter.

3.2 Stream Temperature

Within alpine regions, streams exhibit distinctive thermal regimes due to the influence of snow- and glacier ice-melt; and they are inhabited by a specialised cool water community [32]. Water temperature varies considerably in alpine river systems both temporally and spatially [6, 33–35]. For example, Brown & Hannah [33] showed mean temperature differences of up to 14.2°C across 27 sites in the Taillon basin, French Pyrénées, during the 2003 summer melt season (Fig. 1). Maximum differences in water temperature were even more pronounced, reaching >20°C on some days. The thermal heterogeneity of alpine rivers is influenced by (1) relative contribution from different water sources (including groundwater-surface water interactions), (2) prevailing hydroclimatological conditions, (3) distance from source, (4) total stream flow volume, and (5) basin factors (specifically, valley/channel geomorphology and riparian cover) [28, 34–36].

Alpine meltwater-fed rivers are characterised by large annual temperature fluctuations, with water temperature close to zero or subzero during winter, increasing temperature during spring-early summer, then decreases in temperature during late summer-autumn [31, 35]. In contrast, many alpine groundwater streams are more thermally stable [27, 33, 34]. In summer, diurnal temperature fluctuations of up to 2°C occur for meltwater streams near the glacier margin but, following snowpack melting and uncovering of streams, further downstream diurnal variability can rise to >10°C [6, 36, 37]. Extreme rainfall events during summer typically result in rapid water temperature decreases although occasionally water temperature increases can occur [22].

Downstream increases in water temperature are typical of glacier-fed rivers [34–36] as streams become more exposed to atmospheric heating and less influenced by initial source temperatures. Mean warming rates per km have been reported from between 0.4–0.6°C in alpine basins with glacier cover >30% [34, 35] compared with 7.6°C where glacier cover is only 4% [33]. A key control on warming rates is discharge which directly influences the thermal capacity of the stream. However, longitudinal warming rates can also be modified by tributary inflows from other meltwater sources or groundwater. For example, average meltwater stream temperature decreases of 0.4°C have been reported due to karstic groundwater stream inflows, whereas warmer water sourced from hillslope groundwater streams led to temperature increases (average 1.7–1.8°C; [6]).

3.3 Suspended Sediment

Glaciers are the major source of fluvial-transported fine sediment in alpine rivers with suspended sediment concentrations of >500 mg L^{-1} and turbidity >200NTU reported from some rivers in the European Alps [4, 18]. Suspended sediment concentrations are highest typically in summer when subglacial channels are "flushed" by meltwater passing through a glacier's internal drainage system and entraining sediment eroded and stored during the preceding winter. Distinct diurnal fluctuations are characteristic as a consequence of sediment transport over daily meltwater discharge cycles. Episodically, suspended sediment can derive from re-entrainment of transiently stored sediment along the margins of stream channels during peak flows and heavy precipitation events [38].

Glacier retreat is associated with high runoff and often high sediment yields, which are in part delivered from newly exposed subglacial sediment [39]. Periods of rapid glacier advance may also be associated with high sediment yield [18] as valley train sediments are overridden and reworked by the glacier, a process which also disrupts the subglacial meltwater drainage system.

In contrast to glacier-fed streams, those sourced predominantly from snowpacks transport little suspended sediment [40]. Any sediment in rivers sourced from snowmelt is likely to come from entrainment of river bank sediments [38]. Groundwater streams have suspended sediment concentrations close to zero [27]. However, Tockner et al. [41] describe a predominantly groundwater-fed stream carrying relatively high suspended sediment concentration due to sediment laden glacial water exfiltrating through floodplain alluvial sediments. Clear side-channels sourced from snowmelt or groundwater can also temporarily become turbid when connected to main glacier-fed channels during floodplain expansion cycles associated with flow pulses [8].

3.4 Hydrochemistry

In alpine river basins with glaciers, meltwaters acquire solute from two principal sources: (1) the atmosphere (precipitation and dry deposition) that provides sea salts, acidic aerosols of N and S, plus CO_2 and O_2 to drive chemical weathering reactions; (2) the chemical weathering of rock in subglacial and ice-marginal environments [42]. Hydrochemical characteristics are strongly influenced by seasonal snowmelt, and the development of glacier drainage systems [19, 43].

A distinctive early melt season peak in some solutes occurs due to preferential elution of SO_4^{2-}, NO_3^- and Cl^- from snowpacks [44]. The initial release of acid anions (SO_4^{2-} and NO_3^-) often results in a marked decline in proglacial stream pH (e.g. [45]). Thereafter, melting snowpacks produce waters that dilute more solute rich subglacial runoff and so result in a characteristic inverse relationship between stream discharge and electrical conductivity [46]. Meltwater solute acquisition is enhanced from glacier comminuted rock flour and an abundance of reactive minerals, resulting in waters enriched in carbonates and sulphates [19]. For groundwater streams in alpine river basins, higher solute concentrations are indicative of subterranean water's long residence time [47]. Consequently, alpine groundwater streams are often characterised by relatively high electrical conductivity and enrichment of major ions and silica [5, 6, 27].

3.5 Channel Stability

In alpine streams, channel stability is a major determinant of the abundance and the diversity of the biological communities able to establish [48]. Where bedload transport is high, channel stability is characteristically low. Those streams fed by glaciers tend to have more unstable braided channels which can migrate laterally from year to year. However, if a proglacial lake is present at the margin of the glacier then the channel downstream is more stable as the lake buffers hydrological variation [49].

4 River Ice

Ice and snow is a particular feature of alpine streams, creating unique environmental conditions that strongly affect the flora and fauna both directly and indirectly through changes in their environment. The extent, nature and duration of river ice varies widely across alpine areas, ranging from temporary thin ice cover in low-lying alpine lake outlets and groundwater fed reaches to river channels completely filled by ice.

Several distinct types of ice may form in streams and rivers. In slow flowing, non-turbulent reaches when air temperatures fall below zero during the autumn, a surface ice cover forms, usually first along the shoreline. In the initial phase this layer may move downstream with the current, but as the ice cover becomes more extensive a stationary ice cover develops. As a result of falling discharge and possible melting on the underside due to groundwater inflows, this surface layer may become suspended on boulders. In the more turbulent reaches frazil ice forms when average stream temperatures fall below zero. The flocculation and subsequent growth of frazil particles leads to the formation of ice floes and these may attach to surface ice cover. At low temperatures anchor ice may form on the river bed, either directly or as a result of the stranding of frazil ice.

During winter, water discharged by subsurface flow freezes on reaching the surface, producing sheet-like masses of layered ice, called icings. Icings can block entire river reaches, creating problems when winter snows start to melt in spring, and may cause considerable damage to riparian vegetation. The formation of ice can have profound effects on river hydraulics, changing the channel cross-section, increasing the wetted perimeter, creating new flow pathways, restricting hyporheic flows and adding an additional channel roughness on the underside of the ice cover as well as changing the substrate roughness.

In alpine regions, especially in maritime climates, snow depths may be considerable. This insulates the stream environment from subzero temperatures. River ice is an important environmental component of alpine rivers and has resulted in adaptive mechanisms among the fauna [50, 51]. Despite these adaptations, winter conditions inevitably cause high mortality to stream invertebrates and fish, especially in reaches with unstable snow and ice cover and thereby susceptible to formation of frazil and anchor ice. The lack of winter ice cover in lake outflows and groundwater-fed reaches provides a favourable environment for primary producers and those benthic invertebrates utilising primary production [52].

5 An Integrated Approach: Linking Water Source and Physicochemical Habitat Dynamics

Although we outline individually the major sources of stream flow in alpine environments, it is critical to understand how contributions from these sources change and interact temporally (diurnal to inter-annual) and spatially (longitudinal and lateral). A commonly used approach to conceptualise alpine streams is to classify them using "kryal", "rhithral" and "krenal" as described by Steffan [53] and supplemented by Ward [40]. Kryal streams are considered to be fed by glacial meltwater. Those sourced from snowmelt are termed rhithral and groundwater streams are classed as krenal. However, a modification to this system was proposed by Brown et al. [1] to better reflect the high spatio-temporal variability and interactions among alpine stream physicochemical habitat and biota.

This alternative approach provides a *quantitative* hydrological basis (proportional contribution of each water source) for stream classification and, because alpine streams typically have mixed water sources, increases the number of stream categories from three to nine (Fig. 2). These categories relate water source contributions with physicochemical habitat variables such as stream discharge, suspended sediment concentration and hydrochemistry (Table 1). A key point in developing this system is that over scales ranging from minutes to years and across river networks to reaches, the physiochemical habitat conditions of an individual stream reach can vary considerably because most streams are not fed exclusively by one source. By providing quantitative explanations of the stream "type" as determined by proportions of water sources, this approach seeks to facilitate more accurate comparison between (1) stream reaches and (2) river basins (in space), and (3) changes over time.

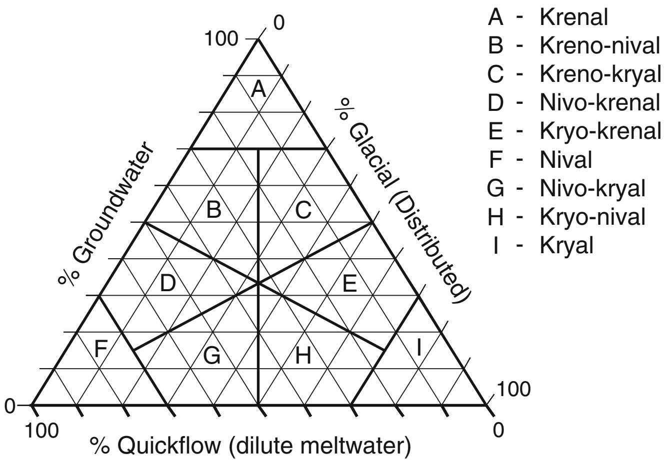

Fig. 2 ARISE categories based on Glacial (Distributed), Quickflow (Dilute meltwater) and Groundwater contributions (see Table 1 for details of categories A–I)

Table 1 Summary of alternative classification categories based on proportions of water sources (modified after Brown et al. [1])

Reference	ARISE category	ARISE category water source contribution descriptors
A	Krenal	High proportion of groundwater (>70%). Combined glacial and quickflow total <30%
B	Kreno-nival	Groundwater sources dominate (35–70%). Quickflow contribution > glacial
C	Kreno-kryal	Groundwater sources dominate (35–70%). Glacial contribution > quickflow
D	Nivo-krenal	Quickflow dominates (35–70%). Groundwater contribution > glacial
E	Kryo-krenal	Glacial sources dominate (35–70%). Groundwater contribution > quickflow
F	Nival	High proportion of quickflow (>70%). Combined glacial and groundwater total <30%
G	Nivo-kryal	Quickflow dominates (35–70%). Glacial contribution > groundwater
H	Kryo-nival	Glacial dominates (35–70%). Quickflow contribution > groundwater
I	Kryal	High proportion of glacial (>70%). Combined quickflow and groundwater total <30%

Recent testing of this new alpine stream classification approach has further highlighted difficulties in separating water sources from water pathways. For example, a large quantity of runoff from glaciers (i.e. traditional kryal classification) is derived hydrologically from both glacial ice- and snow-melt (i.e. kryal and rhithral) during the summer melt season [4, 6]. Additionally, with increasing distance from glaciers, tributary and/or upwelling groundwater inputs (krenal) may supplement meltwater contributions driving further ecosystem changes (e.g. [20]). As such, Brown et al. [6] reconceptualised the contributions of major alpine water sources as quickflow (dilute, rapidly routed snow- and ice-melt), distributed (slow routed meltwater passing through a sub-glacial drainage system) and groundwater (see Table 1). Our new approach to alpine stream classification can be used potentially as a tool for to inform prediction of the hydroecological effects of any future changes in alpine river water sources [3].

6 Alpine Stream Habitat Template

The complex spatio-temporal mosaic of habitats in alpine river systems relates to the interaction of a range of physicochemical processes, with different processes dominating the river environment at different scales. Our new approach to alpine stream classification provides a basis to conceptualise this dynamic habitat template. The contributions from the different water sources have an important influence on the aquatic habitat template for the establishment of biotic communities, particularly through the effect on physicochemical variables outlined above. These include channel stability, water temperature and sediment regimes. Where alpine streams are fed by glaciers, there is a distinct longitudinal pattern of biotic communities with respect to distance from the glacier margin, as outlined in the conceptual model proposed by Milner & Petts [49] and subsequently largely substantiated by Milner et al. [48] see Fig. 3. Although physiochemical habitats in alpine rivers exert a severe constraint on biotic communities, spatial and temporal habitat heterogeneity resulting from these different water sources can enable a diverse flora and fauna to occur [48].

Changes in water sources drive seasonal shifts in benthic community richness, abundance, dispersal and colonisation [52, 54]. For example, reduced densities in the macroinvertebrate and algal communities occur during summer periods of peak glacier-melt [8, 29]. During early spring, snowmelt provides a relatively constant source of water to streams, suspended sediment concentrations are low and elution of nutrients [55] opening a "window of opportunity" [56] to stimulate primary production. Algal growth is typically greatest following snowmelt in spring, leading to increased abundance of grazers (e.g. Diamesinae and Orthocladiinae) [57]. Harsher conditions occur during peak summer melt (i.e. higher suspended sediment concentration, more variable flows and considerable bedload movement) before relatively benign conditions occur as glacial melt decreases into autumn. For example Burgherr et al. [29] found Ephemeroptera and Plecoptera more abundant in the glacier-fed Roseg River, Swiss Alps, during late autumn/early winter when environmental conditions were more favourable due to a greater contribution of groundwater. Harsher conditions due to low temperature and potentially frozen streams return again in winter. More stable year-round patterns in habitat and benthic communities are found in alpine streams where groundwater contributions are high [29, 52, 58].

7 Dominant Biotic Groups in Alpine Streams

Allochthonous inputs into alpine streams are typically low [59]; and thus secondary production is dependent on autochthonous energy sources [60]. Clear longitudinal patterns in algal biomass [61], chlorophyll-a [62] and diatom taxonomic richness [37] occur, related predominantly to changes in water temperature and channel stability. In alpine glacial streams, algal biomass typically peaks during spring when nutrient elution from snowpacks results in the rapid increase in the golden alga, *Hydrurus foetidus* – the dominant alga in alpine glacier-fed rivers [61, 63].

Longitudinal patterns in benthic macroinvertebrate fauna downstream from alpine glacier-fed rivers have been documented in several alpine regions [2, 48, 54, 62, 64–68] as a function of channel stability, water temperature and inputs of allochthonous organic matter (Fig. 3). Diamesinae chironomids, typically dominate near glacier margins where maximum water temperature is $<2°C$ and river channel stability low [62]. A few Orthocladiinae (Chironomidae) can tolerate low stability habitats where maximum water temperature reaches 4°C, and Oligochaeta and Tipulidae can colonize at higher channel stability at these temperatures. Ephemeroptera, Plecoptera and Trichoptera taxa become increasingly abundant further from

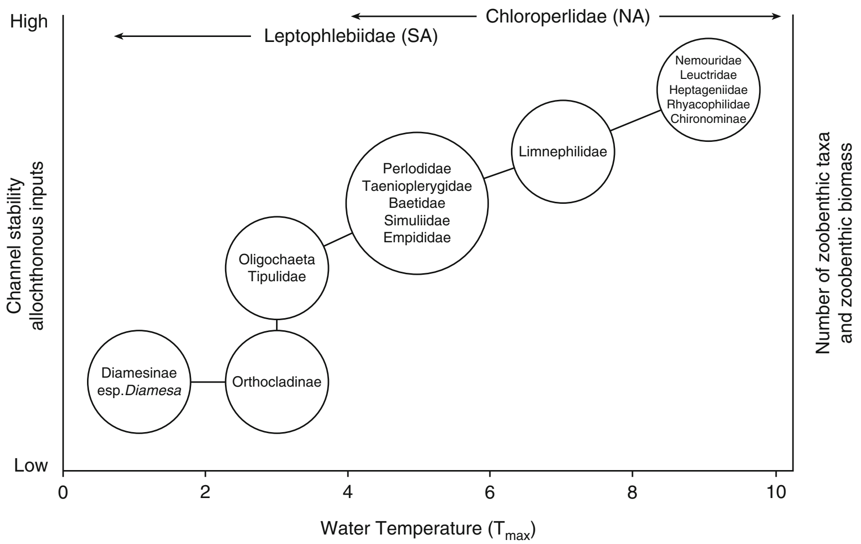

Fig. 3 Conceptual model of the melt season longitudinal ecological patterns of glacial rivers in Europe (adapted from [48])

the glacier snout along with Simuliidae and other chironomids (e.g. Chironominae) as channel stability and maximum water temperature increase. Malard et al. [28, 69] also attributed increases in invertebrate richness with distance from the glacier margin particulate organic matter and upwelling groundwater. Longitudinal changes in species traits are also apparent along glacial rivers, with small, crawling, deposit feeders and scrapers dominating invertebrate communities close to the glacier margin before trait diversity increases downstream [70, 71]. However, alpine glacier-fed streams are influenced by tributary inflows downstream [72, 73]. If the tributary is predominantly glacier-fed then biotic diversity is likely to decrease and if predominantly groundwater then biotic diversity will potentially increase.

The greater channel stability and less variable water temperature regimes in krenal streams typically results in greater macroinvertebrate diversity. For example, Klein & Tockner [74] identified five major groups of macroinvertebrates (Heteroptera, Coleoptera, Gastropoda, Bivalvia and Ostracoda) almost entirely restricted to groundwater-fed alpine streams on the Val Roseg floodplain. In the Austrian Alps found 30 taxa restricted to a spring-fed system and absent from a nearby glacier fed system [30]. Similarly, of five endemic Pyrénéan species found in the Taillon-Gabiétous basin, French Pyrénées, four were principally associated with groundwater streams [71]. However, although biodiversity is typically higher in the groundwater systems, stream community stability (the similarity of community composition over time with respect to relative abundance) is often highest in glacier-fed streams. Brown et al. [75] suggests that sites with more stable physicochemical habitat variables (e.g. groundwater-fed streams) may not necessarily support the most stable communities, possibly due to species abundance patterns being influenced more by changes in resource availability and biotic interactions than physical determinants [76].

8 Summary

We have attempted to provide an overview of the major driving variables that serve as a habitat template for biotic communities in alpine streams. It is evident that these are dynamic systems with considerable spatial and temporal variation in a number of key variables downstream of their source. The temporal changes in water sources during the melt season have a major influence on river discharge, sediment transport, solute chemistry, water temperature, channel stability and primary producers. These temporal changes in water sources are sensitive to anthropogenic pressures including climate change, water resource allocations, and contaminants which will ultimately influence water quality and habitat suitability for biotic communities.

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Biodiversity of Flora and Fauna in Alpine Waters

C.T. Robinson, B. Kawecka, L. Füreder, and A. Peter

Abstract The study of Alpine aquatic environments began in the early 19th century, but only since the early 1990s has scientific interest intensified on these systems. The goal of this chapter is to summarize the community patterns of algae, zoobenthos, and fish that occur in Alpine freshwaters. Benthic algae in Alpine waters are differentiated among particular regions of the Alps in relation to geology, stream origin, and anthropogenic activity. Because of their dominance, the diatoms are the most widely studied algae in Alpine waters. By possessing various morphological structures and physiological traits, most stream insects are adapted to the dynamic and cold aquatic habitat of alpine landscapes. Aquatic insects comprise a substantial proportion of the zoobenthos in surface waters, with Chironomidae being most common. Seasonality is a common feature of macroinvertebrate assemblages in glacial streams in the Swiss Alps. Two evolutionarily successful strategies in glacial streams are adaptation to unstable stream conditions during summer (summer species) or avoidance of these conditions (winter species). Only a few

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native fishes are able to colonize and inhabit Alpine waters. Over the last decades, non-native, cold stenothermic species have established self-reproducing populations and appear well-adapted to the harsh environmental conditions in the Alps. The indigenous brown trout (*Salmo trutta fario* L.) is the most important fish in alpine running waters. Glacier retreat has accelerated globally, increasing the probability that fundamental ecological changes will occur in alpine landscapes, in particular the ecology of running and standing waters.

Keywords Alpine diatoms, Alpine fishes, Alpine macroinvertebrates, Environmental indicator, Glacier retreat

Contents

1 Introduction

The study of Alpine aquatic environments began in the early 19th century, but only since the early 1990s has scientific interest intensified on these freshwater habitats. Most of these studies have been conducted in the Southern Alps, Swiss Alps and Central Austrian Alps, and included lakes, springs and streams above 2,000 m asl.

In the Southern Alps (Italy), most studies were done in natural parks in the province of Trentino such as Adamello-Brenta Regional Park, including the Adamello range on crystalline bedrock and the Brenta massif with bedrock of sedimentary limestone and dolomite. Here, the Malga Nambi and Borzago were extensively examined. Another area was the Ortles massif that includes Stelvio National Park with its snow-melt and glacial-melt streams (e.g. Plima, Furkel, and Madriccio). In the Swiss Alps, headwater streams of the Rhone, Rhine, Po and Danube Rivers have been examined, including lake outlets, snow-fed (rhithral), glacial-fed (kryal), and groundwater (krenal) streams. Much work has been completed in the Swiss National Park near the Italian border, especially the Spöl River that drains Livigno Reservoir. Here, the Macun lakes region is a recent annex to the park and is designated for long-term monitoring of alpine freshwaters. The Val Roseg in the upper Engadine and Mutt Glacier in the Rhone basin also have been extensively studied. Various lakes in French and Italien Alps have been examined in the framework of different EU Projects such as ALPE- Alpine Lakes: Paleolimnology and Ecology, MOLAR- Mountain Lake Research, and EMERGE- European Mountain Lake Ecosystems.

In the Austrian Central Alps (Tyrol), the Rotmoos within the UNESCO Biosphere Reserve in the Ötzal range has been an area of extensive study, including the streams Rotmoosache, Gurgler Ache, Königsbach and Radurschlbach (each having a different water source). In the Stubaier range, streams draining the former lakes Hintertaler and Vorderer Finstertaler See, and also Gschnitzbach have been investigated. The Isar River in the northern Calcareous Alps has been a topic of ecological research for many years as well as Warme Mandling spring in the Dachstein range. Lastly, a number of calcium-rich and calcium-poor streams have been studied in the eastern Alps.

Most natural Alpine waters are nutrient limited with nutrients entering during the seasonal flow pulse from snow-melt and glacial-melt waters [1]. Nitrogen content tends to be high because of atmospheric deposition, typically being $>200\ \mu g\ L^{-1}$ nitrate-N in flowing waters. Temperatures of Alpine waters typically range between 0.1 and 18°C, being substantially colder in kryal streams. Waters situated on crystalline bedrock have pH's of 5.4–7.6 and conductivities of 4–135 $\mu S\ cm^{-1}$, whereas those on sedimentary bedrock typically have pH's of 7.1–8.5 and conductivities of 141–302 $\mu S\ cm^{-1}$. Nutrient enrichment impacts a number of Alpine waters such as the Gurgler Ache, Finstertaler and Warme Mandling streams influenced by the tourist centres of Obergurgler, Kühtai and Filzmoss, respectively. Here, phosphate can reach 6–16 $mg\ L^{-1}$, total phosphorus 10.7 $mg\ L^{-1}$ and total nitrogen 28.2 $mg\ L^{-1}$ [2, 3]. Water regulation and abstraction impacts numerous Alpine streams such as the Plima, Spöl and Radurschlbach. The Spöl River now has an artificial flood programme in place to improve ecological conditions [4]. Alpine waters are also threatened by airborne pollutants causing acidification, heavy metal contamination, pesticide loading and nutrient enrichment [5, 6]. Alpine waters are well-known to be sensitive indicators of landscape transformation such as from climate change and anthropogenic stressors [7, 8].

The goal of this chapter is to summarize the composition and community patterns of algae (with special consideration to benthic diatoms), zoobenthos, and fish that occur in Alpine freshwaters. The data presented are a synthesis of published research and represent the current state of knowledge on the different groups. The chapter closes with a general perspective on the biodiversity of alpine freshwaters.

2 Algae of Alpine Waters

Benthic algae in Alpine waters are differentiated among particular regions of the Alps in relation to geology, stream origin, and anthropogenic activity. In general, most waters are dominated by diatoms. For example, in the Rotmoos glacial stream in Austria, of 278 phytobenthos taxa, 181 species were diatoms, 42 were cyanobacteria, and 40 were chlorophyceae [9]. Oligotraphentic and acidophilous diatoms prevail in waters flowing over crystalline bedrock and alkaliphilous groups in waters over limestone bedrock. Diatom communities are mostly influenced by water pH, degree of mineralization, sulphates, nutrients, and discharge. Glacier meltwater has an especially strong effect on algal composition. Seasonality is pronounced in some alpine environments and strongly controls algal growth and productivity. The most frequently occurring diatom in Alpine waters is *Achnanthidium minutissimum*, similar to other high elevation regions on the globe. In Alpine waters, many species inhabit surface waters as well as moist areas. There are also many endangered and even rare species placed on the German Red List [10].

Because of their dominance, the diatoms are the most widely studied algae in Alpine waters [1, 3, 9, 11–22] (Fig. 1). Benthic diatoms inhabit bedrock and mosses, and grow on cyanobacteria (mainly *Homoeothrix janthina, Phormidium* spp.), gold algae (*Hydrurus foetidus*), and green algae (*Ulothrix zonata, Klebsormidium rivulare*) that are often found in Alpine streams [1, 3, 9, 12, 23–30]. Diatoms are excellent environmental indicators and often used to monitor environmental changes in most kinds of waters [31–34].

2.1 Diatom Communities in Natural Waters

2.1.1 Lakes

Benthic algal assemblages of alpine lakes have been relatively little studied. Epilithic diatoms in Alpine French lakes were dominated by *Achnanthidium minutissimum* and *Denticula tenuis*, and in Italian lakes also by *Achnanthes scotica*, *Psammothidium helveticum, P. marginulatum*, *P. subatomoides*, and *Eunotia exigua* (see Wathne et al. [35, 36]). The species richness of littoral diatoms of Adamello-Brenta Regional Park (Italy) and the Macun Lakes area in the Swiss National Park was relatively high (204 and 109 species, respectively). Both areas had a similar number of genera (29/28) with *Navicula*, *Eunotia, Pinnularia,*

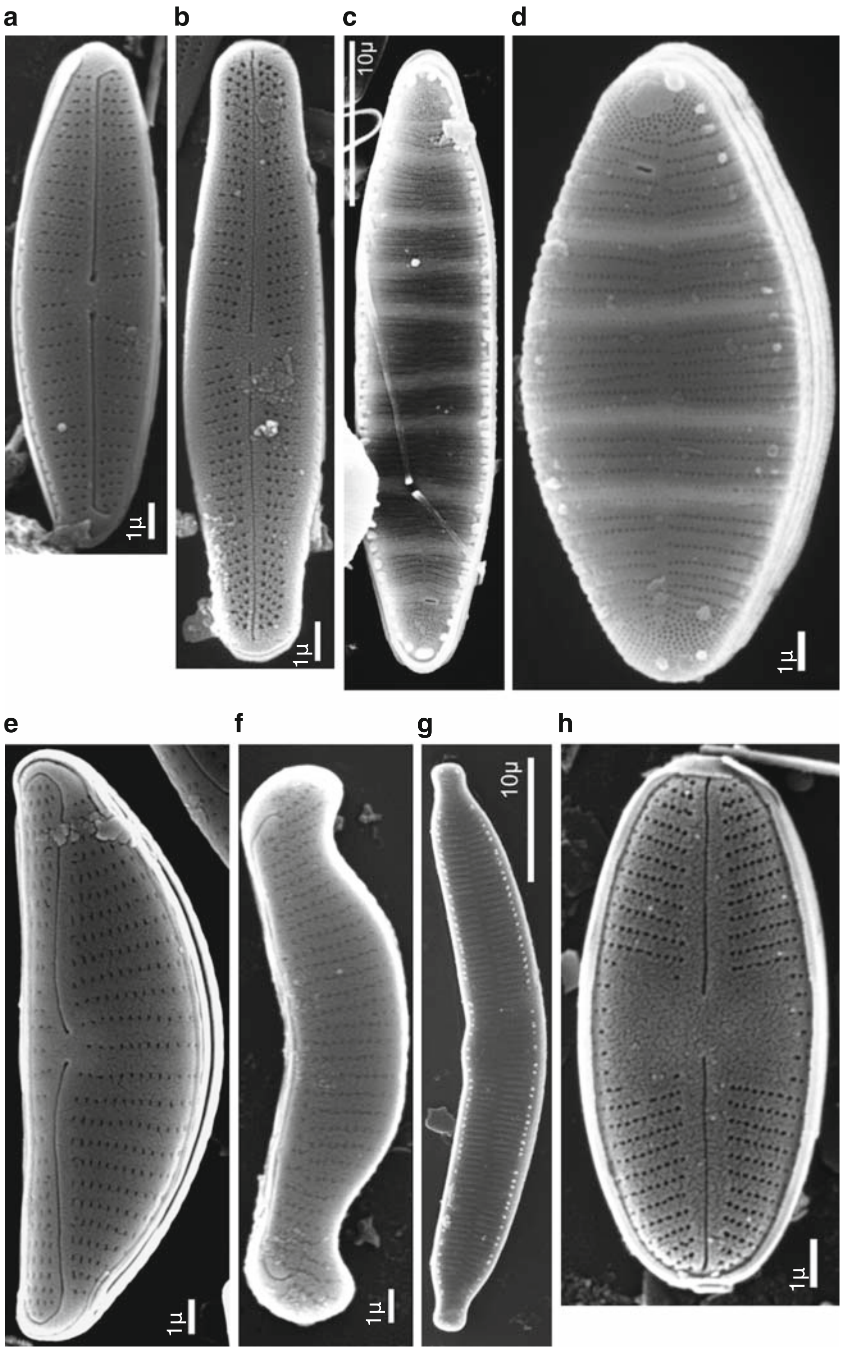

Fig. 1 The most common Alpine diatoms shown by scanning electron microscopy. Not marked scales mean 1 µm. (a) *Achnanthidium biasolettianum* (Grunow) Round & Bukhtiyarova (9,400× magnification); (b) *Achnanthidium minutissimum* (Kützing) Czarnecki (7,800× magn.);

Achnanthes and *Cymbella* being the most common. In Adamello-Brenta, the most frequently occurring species were *Psammothidium marginulatum, Achnanthidium minutissimum, Psammothidium acidoclinatum, P. helveticum* and *Brachysira brebissonii*. Among these, *Achnantidium minutisimum* was most abundant. The Macun Lakes area is composed of two basins with different water sources: snow-melt and glacial-melt. Lakes in the northern snow-melt basin had higher abundances of *Aulacoseira alpigena, Achnanthidium minutissimum* and *Aulacoseira distans* than the south basin lakes. In the glacial-fed south basin lakes, as in Adamello-Brenta, the most common diatoms were *Psammothidium marginulatum, P. helveticum, P. subatomoides* and *Pinnularia microstauron* [6, 19, 21, 22].

2.1.2 Springs and Streams

In springs of Adamello-Brenta Regional Park in Trentino, Italy, 250 diatom species were found that displayed similarities to diatom communities of springs and headwaters of other geographic areas [14]. Diatom communities were clearly related to a gradient in pH and arranged into a few groups. The most frequent and abundant diatoms were *Achnanthidium minutissimum* and *Diatoma mesodon*. Two distinctly different groups were observed: those in siliceous springs with characteristic acidobiontic (*Eunotia subarcuatoides, E. exigua*) and acidophilous (*Brachysira brebissonii, B. neoexilis, Eunotia implicata, Gomphonema amoenum, Psammothidium marginulatum)* species as well as species of boreal-alpine distribution, and those of calcareous springs with a uniform species composition dominated mainly by alkaliphilous species of *Achnanthidium biasolettianum, Encyonema minutum, Gomphonema pumilum, Meridion circulare* and *Planothidium lanceolatum.*

In the Natural Parks of Trentino (Adamello, Brenta, Ortles), several groups of streams and springs arranged along a conductivity and alkalinity gradient were recognized [18]. For instance, the group of streams and springs associated with low conductivity and alkalinity had characteristic acidobiontic and acidophilous species: *Psammothidium acidoclinatum, Eunotia subarcuatoides, E. exigua, Gomphonema amoenum, Fragilaria capucina* var *rumpens* and *P. marginulatum.* In streams with intermediate conductivities and reduced alkalinities, common taxa were *Diatoma mesodon, D. hyemalis, Fragilaria arcus, F. virescens, Gomphonema parvulum, G. exillisimum,* and *Nitzschia alpina.* The group of springs with high conductivity, alkalinity and pH were inhabited by species characteristic of mountain streams of mid-altitude, such as *Achnanthidium minutissimum, A. biasolettianum*, as well as *Achnanthes conspicua*, *Amphora pediculus, Denticula tenuis, Planothidium lanceolatum* and *Reimeria sinuata*.

(c) *Diatoma hyemalis* (Roth) Heiberg; (d) *Diatoma mesodon* (Ehrenberg) Kützing (6,600× magn.); (e) *Encyonema minutum* (Hilse) D.G. Mann (6,600× magn.); (f) *Eunotia exigua* (Brébisson) Rabenhorst (7,200× magn.); (g) *Fragilaria arcus* (Ehrenberg) Cleve (2,400× magn.); (h) *Psammothidium marginulatum* (Grunow) Bukhtiyarova & Round (9,400× magn.)

Diatoms are poorly represented in groundwater-fed (krenal) streams in the north-central Alps. *Achnanthidium minutissimum, Diatoma mesodon* and *Fragilaria arcus* were most common in krenal streams in the Rotmoos catchment, whereas *Encyonema minutum, Diatoma hyemalis, Gomphonema angustatum* and *Meridion circulare* were dominant in Madriccio ([9], Kawecka, unpublished data). In contrast, diatoms were better represented in krenal streams in the southern Alps. For instance, 94 species were recorded from the Borzago and Malga Nambi with differences between streams due to differences in lithology, pH, conductivity, alkalinity and sulphate. Species richness increased and composition changed longitudinally along the Borzago, which flows over siliceous bedrock, with *Cocconeis placentula* var *euglypta, Achnanthidium minutissimum* and *Gomphonema pumilum* sensu lato becoming more common. In Malga Nambi, which flows over carbonate bedrock, the diatom community was exclusively dominated by *Achnanthidium biasolettianum* and *A. minutissimum* [16, 17].

Glacier (kryal) streams are a common feature in alpine landscapes. The relatively harsh environment of glacier streams is reflected in their low species richness and abundance of diatoms. For instance, Hieber et al. [28] recorded only about 10 diatom genera from glacial-fed streams in the Swiss National Park. The most abundant genera were *Achnanthes*, *Cymbella*, *Diatoma* and *Gomphonema* and were similar to those found in the Rotmoosache glacial stream [9]. Species richness ranged from 18 species in the stream Val d'Aqua to 63 in the Rotmoosache [9, 11, 12, 20, 28]. The most common species were *Achnanthidium minutissimum*, *Diatoma hyemalis, D. mesodon, Encyonema minutum, Fragilaria arcus, Gomphonema angustatum, G. angustum* and *Tabellaria flocculosa* [9, 12, 18, 20].

2.1.3 Lake Outlets

Environmental conditions tend to be ameliorated in lake outlet streams, making them conducive habitats for algal development. However, distinct differences in diatom composition were noted between different Alpine basins. Lake outlets in the Swiss Alps had 16–27 genera and 20–46 species [28]. Here, *Achnanthidium minutissimum, Encyonema minutum, Fragilaria arcus, Fragilaria delicatissima* and *Fragilaria capucina* var *rumpens* dominated. In outlets of the Macun Lakes, 39 genera and 109 species were recorded in the rhithral northern basin and 34 genera and 108 species in the kryal southern basin. A total of 143 diatoms was recorded for both basins. *Navicula, Fragilaria, Eunotia* and *Pinnularia* had the highest number of species. The most common species in both basins were *Achnanthidium minutissimum*, *Gomphonema parvulum, Fragilaria capucina* and *Pinnularia sinistra. Aulacoseira alpigena* had higher abundances in the north basin, whereas *Psammothidium helveticum, P. marginulatum, P. subatomoides* and *Diatoma mesodon* were more abundant in the south basin [21, 22]. Between 60 and 77 diatom species were recorded in the rhithral outlets of Lakes Hinterter and Schwenzer in the Austrian Alps. *Diatoma hyemalis* and *D. mesodon* dominated communities (large masses occurred in Königsbach stream), although *Achnanthidium minutissimum* and *Encyonema minutum* also were common [12].

2.2 Diatom Communities in Alpine Waters Impacted by Man

2.2.1 Eutrophication

Alpine streams near dense tourist areas, such as Gurgler-Ache, Finstertaler and Warme Mandling (Obergurler, Küthai, Filzmos, respectively), are usually threatened by eutrophication [3, 11, 12]. Downstream of human settlements, the structure of diatom communities indicates nutrient-enriched environments. The number of diatoms and their abundances increase, and most of these display a wide trophic spectrum or preference for nutrient-rich waters. For instance, *Encyonema minutum* (oligo-beta mesotraphentic [37]) had high abundances and *Navicula cryptocephala* (oligo-eutraphentic) and *Nitzschia palea* (hypereutraphenthic [38]) were common in the Gurgler–Ache glacial stream. Similar diatom communities developed in the Rybi Potok stream (Tatra Mts, Poland) affected by a tourist shelter [12, 34, 39]. In the nutrient-enriched Finstertaler stream, species richness reached as high as 122 taxa, being dominated by *Fragilaria arcus* (oligo-mesotrophic) and *Encyonema minutum*. A considerable increase in the abundance of *Encyonema minutum* and species of *Gomphonema* and *Nitzschia* were recorded in Warme Mandling stream downstream of a sewage treatment plant.

Alpine waters are also threatened by airborne pollutants causing acidification. Marchetto et al. [5, 40] documented acidification problems in lakes of the Central and Southern Alps (also see [35, 36, 41, 42]). Especially endangered by acidification are poorly mineralized waters situated on crystalline bedrock [40]. Appearing in springs, *Eunotia subarcuatoides* and *E. exigua* are described as species typical of acidified running waters [14, 43]. In contrast, Tolotti [19] found that although lakes are sensitive to acid inputs, pronounced acidification in lakes of the Italian Alps could not be detected. In the Macun Lakes (Swiss Alps) acid inputs were also of minor influence on epilithic diatom communities [6].

2.2.2 Flow Regulation

Flow regulation and water abstraction are serious ecological threats to Alpine streams and rivers. Algal communities often shift in composition following changes in flow regime [13]. Above and directly below a reservoir on the Plima stream, diatom richness was low and was dominated by species of *Achnanthidium minutissimum, Encyonema minutum, Cymbella affinis, Diatoma hyemalis* and *D. mesodon*. A more drastic change was observed lower in the stream below the town of Martello in which a major increase in the diatom community was recorded, mainly via increases in *Achnanthidium minutissimum, Encyonema minutum, Gomphonema* *angustatum* and *G. olivaceum*: changes attributed to the influence of biogens (Kawecka, unpubl. data).

In the Spöl River below Livigno reservoir, the diatom community was rich with 134 species. *Achnanthidium minutissimum* and *A. biasolettianum* dominated the community and *Gomphonema angustum, Encyonema minutum, Fragilaria capucina austriaca* group, *Diatoma ehrenbergii* and *Cocconeis placentula* var *euglypta* also were quite abundant. Flow regulation of the Spöl contributed significantly to the high algal biomass relative to nearby streams having a natural flow. Introduced experimental floods reduced the number of diatom species and algal biomass, and changed the diatom community composition [20, 44].

2.3 Factors Influencing Diatom Community Structure

Most studies of undisturbed aquatic habitats in the Alps showed that diatom community structure was mainly determined by geochemical factors such as pH, alkalinity, degree of mineralization, sulphate content as well as physical conditions such as flow regime and current velocity [9, 14, 17–19]. Tolotti [19] noted that the influence of pH is a particularly significant determinant of diatom communities in lakes, thus supporting the results of studies carried out in other mountain areas of Europe (Italy, Switzerland, Austria, Scandinavia). A major factor governing the community structure among stream types was the presence of a glacier. Glacier melt causes strong seasonal fluctuations in discharge, which in turn, determines a stream's physico-chemistry (e.g. temperature, turbidity, nutrient availability).

In kryal streams, the most favourable period for algal growth is spring and late autumn with very low abundances found in summer [1, 28]. Upstream lakes are another important determinant of benthic algal communities. Lakes generally buffer temporal fluctuations in environmental conditions of their outlets, thereby causing some degree of habitat stability for organisms during the year. Algal communities in lake outlets show little seasonal change in species richness and abundance, as also found in lakes and springs. Rhithral streams show little seasonality because the growth period is relatively short (ca. 4 months) in alpine landscapes [14, 19, 28]. In contrast, krenal streams display strong seasonality, shifting from a late spring/summer community to an autumn/winter community that coincides with the onset of discharge and nutrient peaks that persist through winter and into early spring [16].

2.4 Diatoms as Indicators of Environmental Conditions

The most frequently occurring diatom in Alpine freshwaters is *Achnanthidium minutissimum*. This species is also abundant in other high-altitude regions of Europe, Himalayas, Andes, Rocky Mountains and Lapland [18, 34, 45–47]. It is a circumneutral species and has a wide trophic spectrum, although requiring oxygen saturated waters [38]. It is known as an early colonizer of open habitats and seems highly resistant to scour [48–50]. Waters of the Alps also had an abundance of *Diatoma mesodon, Fragilaria arcus, Encyonema minutum, Reimeria sinuata* and *Staurosira pinnata* [1]. These species are widely distributed and abundant in other high-altitude areas of Europe and Himalayas [18, 34].

Ecological groups of algae (according to Van Dam et al. [38]) reflect the character of the environment. Oligotraphentic algae are most common in Alpine waters, ranging from 26% of the species composition in the Rotmoosache glacial stream [9] to 39% in springs [14]. Acidophilous diatoms with an optimum pH between 5.4 and 6.9 are common in lakes over crystalline bedrock [19]. In streams in the Natural Parks of Trentino that flow over heterogeneous geology, diatom communities consist mainly of circumneutral (38%), alkaliphilous (30%) and acidophilous (25%) species [18].

A large group of diatoms is adapted to desiccation. On 50% of the recorded diatoms in the Natural Parks of Trentino were associated both with water bodies and wet places, and 18% mainly occurred in wet places [18]. *Diadesmis gallica* var *perpusila* was often found in Alpine streams and springs [14], a species known to occur nearly exclusively also outside of water bodies [38]. Some diatoms showing a narrower range in distribution included species of Nordic-Alpine preference. Tolotti [19] noted about 60% of the species in Alpine lakes were “mountain-alpine” or “northern” taxa. The most frequently recorded species having a Nordic-Alpine preference (according to Kramer and Lange-Bertalot [51–54]) included *Psammothidium marginulatum, P. subatomoides, P. chlidanos, Aulacoseira alpigena* and *Nitzschia alpina* [14, 18, 19, 21, 22].

A number of Alpine diatoms are on the German Red List and are classified as endangered to various degrees or as extremely rare [10]. These species represented 12% of the diatom community in the Rotmoosache glacial stream [9], 48% in the streams of the Natural Parks of Trentino [18] and 50% in the lakes studied by Tolotti [19]. Most of these species were sporadic in abundance, but the most frequently encountered were classified as endangered (category 3). These included *Achnanthes metakryophila, P. marginulatum, Amphora inariensis, Gomphonema amoenum, Psammothidium chlidanos* and *Eunotia praerupta* [14, 18]. Interestingly, *Gomphonema amoenum* occurs in low-conductivity Alpine waters typical of crystalline areas and seems to be restricted to Europe [18].

Many critically and strongly endangered diatoms were mostly found in streams with siliceous substrate [9, 18, 34]. In the almost extinct group (category 1) was included *Didymosphenia geminata*. It was sporadic in number in most Alpine streams but numerous in the Spöl River in 2001. In fact, this species was previously known as a diatom inhabiting ultra-oligotrophic waters, but now is expanding and developing high abundances in nutrient-enriched streams and rivers of Europe and other parts of the world [55, 56]. Diatoms classified as endangered and extremely rare in other high elevation regions comprised ~33% of the algal community in the Tatras, 21% in the Fogaras, 17% in the Rila Mts, and 34% in the Himalayas [18, 34].

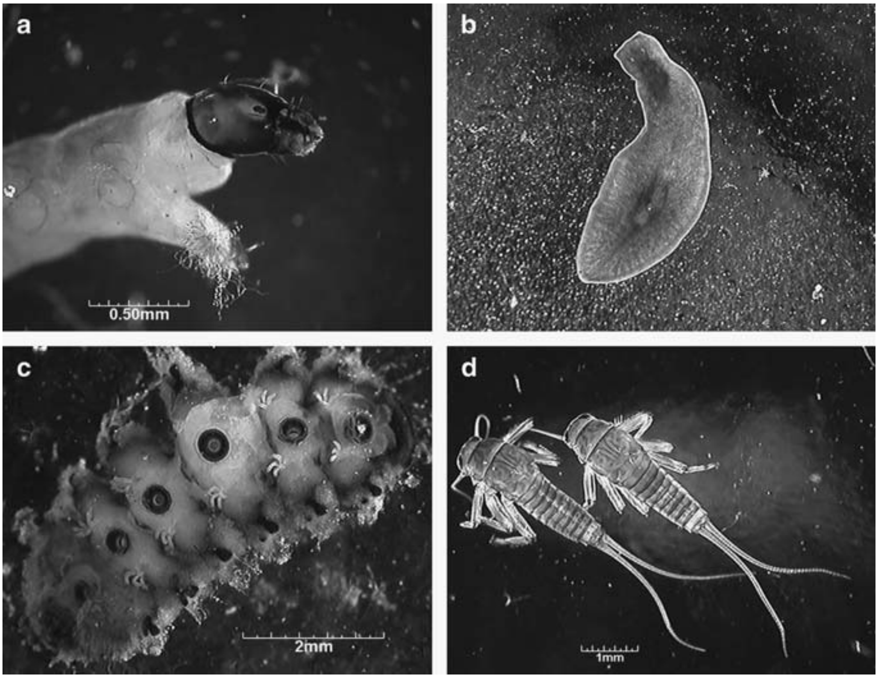

Fig. 2 Some common macroinvertebrates found in high Alpine streams. (a) Chironomidae, *Diamesa*, (b) Turbellaria, *Crenobia alpina*, (c) Blephariceridae, and (d) Ephemeroptera, *Baetis alpinus*

3 Zoobenthos of Alpine Waters

This section begins with a case study on Rotmoosache and Königsbach streams in Austria, and is followed by some general ecological observations of macroinvertebrates in Alpine streams. An investigation comparing the glacier-fed Rotmoosache with the spring-fed Königsbach on a monthly basis over two years yielded 81,385 specimens from the two streams [57, 58]. A total of 126 aquatic or semi-aquatic taxa, representing 13 families from nine orders (Diptera, Ephemeroptera, Plecoptera, Trichoptera, Hydracarina, Crustacea, Oligochaeta, Turbellaria, Nematoda) were collected by benthic sampling and emergence traps for adults (Fig. 2). Ninety-four taxa were found in the glacier-fed stream compared with 120 taxa in the spring-fed stream, of which four taxa were present only in the glacier-fed stream and 28 only in the spring-fed stream.

Aquatic insect abundance and taxon richness was lower in the glacier-fed Rotmoosache (mean 2,798 m^{-2}; 72 taxa) compared with the spring-fed Königsbach (mean 5,357 m^{-2}; 93 taxa). An average emergence rate of 139 ind m^{-2} day^{-1} in Rotmoosache relative to 706 ind m^{-2} day^{-1} in Königsbach underlined this type-specific difference. Chironomids clearly dominated insect emergence in both

Rotmoosache (annual mean emergence rate of 135 ind m^{-2} day^{-1}) and Königsbach (655 ind m^{-2} day^{-1}), whereas EPT taxa (Ephemeroptera, Plecoptera and Trichoptera) had relatively low average emergence rates of 4 and 51 ind m^{-2} day^{-1}, respectively. Chironomid larvae also dominated the benthos, averaging 67% of total macroinvertebrate abundance in Rotmoosache and 39% in Königsbach. Among emerging taxa, they contributed >90% of the total numbers in both streams. The Diamesinae dominated in the glacier-fed stream, making up 65% of the total invertebrate abundance, but accounted for only 11% in the spring-fed stream. Conversely, Orthocladiinae were more abundant in Königsbach (26%) than in Rotmoosache (2%).

Dominant taxa in the glacier-fed stream were larvae of *Baetis* (Ephemeroptera) and *Diamesa* with a relative abundance >10% and 5–10%, respectively. *Rhithrogena*, *Diamesa cinerella*-gr. and *D. latitarsis*-gr. had relative abundances between 2 and 5% in the benthos. Within the emerging EPT taxa, *Capnia vidua* Pictet and *Acrophylax zerberus* Brauer were dominant, accompanied by *Leuctra rosinae* Kempny, *Protonemura lateralis* Pictet and *Dictyogenus alpinum/fontium* Pictet. Among the chironomids, *D. bertrami* Edwards and *D. latitarsis* Goethgebuer each reached >10% of the assemblage, while *Eukiefferiella tirolensis* Goethgebuer and *Stilocladius montanus* Rossaro accounted for 5–10%.

The benthic fauna of the spring-fed stream was dominated by larval Ephemeroptera (*Baetis* and *Rhithrogena*), along with *D. latitarsis*-gr., *Orthocladius rivicola*-gr., Drusinae and crustaceans. The dominant emerging species within the EPT taxa were *Baetis alpinus* Pictet, *Rhithrogena loyolea* Navas, *L. rosinae*, *Consorophylax consors* McLachlan, *Drusus discolor* Rambur and *D. melanchaetes* McLachlan, although all species occurred in low numbers compared with chironomids. Although *Diamesinae* were, in general, less important in Königsbach than in Rotmoosache, *D. latitarsis* was a dominant species. *Corynoneura lobata* Edwards, *Heleniella dorieri* Serra-Tosio, and *Krenosmittia boreoalpina* Goethgebueri were the dominant Orthocladiinae.

3.1 Longitudinal and Seasonal Trends Within Chironomidae

There was a distinct longitudinal succession of chironomids in the glacier-fed stream. Near the glacier, the chironomid assemblage consisted of four taxa: *D. steinboecki* Goethgebueri, *D. cinerella-zernyi*-gr., and larval Diamesinae and Orthocladiinae. However, apart from *D. steinboecki*, these taxa as well as *D. latitarsis*-gr. had the highest relative densities at three downstream sites. Although taxon richness and abundance were more uniform along the longitudinal gradient in Königsbach, species composition changed downstream in both streams.

Seasonal patterns in taxon richness were also different in the two streams. For example, during snowmelt and in summer, Diamesinae and Orthocladiinae richness was higher in Königsbach, whereas in autumn and winter the glacier-fed stream had more taxa within the two subfamilies. In both streams, taxon richness was highest in summer at all sites. Taxon richness in the glacier-fed stream generally decreased in autumn and winter, but at the lower site it increased again in winter to a maximum of 27 taxa. Seasonally, chironomid abundance was similar in Königsbach (winter mean 1,598 ind m^{-2}, summer mean 2,635 ind m^{-2}). In Rotmoosache, in contrast, the winter mean of 3,841 ind m^{-2} was about 5–10 times higher than values for the rest of the year (mean 616 ind m^{-2}). The average density in Rotmoosache during winter was even higher than the summer mean in Königsbach. Although Diamesinae reached an average density of 1,621 ind m^{-2} during summer, their abundance declined to $<2\%$ of this value in autumn and winter. In the glacier-fed Rotmoosache, the mean chironomid richness (3–4), Shannon diversity index (0.86–0.94) and evenness (0.69–0.77) were uniformly low in all seasons and never exceeded the corresponding values for the spring-fed stream.

3.2 Chironomid Diversity, Feeding Groups and Zonation Indices

Distinctly different patterns in chironomid diversity were evident in the two streams. In Rotmoosache, a generally low taxon richness increased with higher abundance ($r = 0.38$) but evenness showed a weaker relationship ($r = 0.30$). In contrast, in Königsbach, the chironomid abundance was more uniformly distributed among the samples regardless of taxon richness, although evenness values increased with increasing taxon richness ($r = 0.57$). Diversity (Shannon Index) was more related to evenness in the spring-fed stream than in the glacier-fed stream.

Comparisons of “species trait” indices in chironomid assemblages of the two streams showed differences in terms of the potential use of nutrients (Saprobic Index), functional feeding groups and the longitudinal distribution of the fauna. Whereas chironomids characteristic of xeno-saprobic and oligo-saprobic conditions dominated the glacier-fed stream (62%), more taxa indicating b-meso-saprobic conditions (47%) occurred in Königsbach. Grazers and detritivores dominated in both streams, while predators were of some importance in Rotmoosache as were predators, filter feeders and shredders in Königsbach. The assignment of longitudinal zonation indices for the chironomid fauna showed a clear dominance of kryal taxa in the Rotmoosache, whereas a broader allocation to stream zonation categories was found in the Königsbach.

3.3 Longitudinal and Seasonal Patterns

Our findings generally support the conceptual model of Milner and Petts [59]. Chironomid and overall taxon richness and abundance increased with increasing distance from the glacier. Diamesinae were the dominant group with *D. steinboecki* being the most abundant species and *D. cinerella-zernyi*-gr. being co-dominant at the glacial snout. Apart from a few larval Orthocladiinae, several non-chironomid taxa were also found at the glacial snout: Capniidae/Leuctridae, Perlodidae, *Baetis* and *Rhithrogena* species, and the aquatic coleopteran *Helophorus glacialis* Villa [60]. Similar findings were reported by Bretschko [61] for several Austrian glacier-fed streams, where *D. steinboecki* was found in association with the plecopteran *Rhabdiopteryx alpina* Kühtreiber, the tipulid *Trimicra* sp., *Baetis alpinus* and nine other taxa. Records of several abundant taxa near the glacier snout are common. A richer fauna than one would expect based on the model by Milner & Petts [59] may be due to the influence of diverse water sources even at the glacial snout. Besides glacial meltwater, groundwater, stream water from higher altitudes entering the glacier, and water arising from snowmelt and precipitation can contribute to the total discharge, creating a more benign environment.

Longitudinal changes in faunal composition are more likely to be found where strong environmental gradients occur [62]. The reduction in taxon richness with increasing altitude reflects distinct physico-chemical changes in the glacier-fed stream. Spring-fed streams are expected to display fewer longitudinal differences because of little change in environmental conditions downstream. This is in accordance with our findings in Königsbach. Season also appears to have a strong effect on invertebrate assemblages in high alpine regions. For example, Schütz et al. [63] found that the importance of several environmental variables explaining faunistic patterns varied across the year. Discharge and maximum temperature were important in summer, while nutrient availability and daily temperature amplitude were important in all seasons. Although in the study above, the average abundance, species diversity and taxon richness in the glacier-fed stream were low, values of single samples and seasonal means were relatively high and resulted in a clear increase in average abundance and richness in winter (also see [63]). Further, in the glacier-fed stream, the chironomid assemblage was different in winter relative to the rest of the year.

The overall input of allochthonous organic matter to high Alpine streams is low [64], as these headwaters drain catchments where terrestrial vegetation is scarce or lacking [59, 65]. However, enhanced food availability may promote invertebrate communities during winter. In a comparison of food availability in glacier-fed versus spring-fed systems, Füreder et al. [66, 67] showed relatively high BPOM and chlorophyll a values in epilithic biofilms. At low discharge and high channel stability in winter, heterotrophic micro-organisms and, in ice-free reaches, primary producers (e.g. diatoms and *Hydrurus* mats) can benefit from available nutrients and form epilithic biofilms subsequently consumed by grazers. The shift in winter from species indicating xeno- and oligotrophic conditions to taxa typical of higher saprobity levels and the marked increase in chironomid abundance in lower reaches suggest better food conditions at this time. During winter, glacier-fed streams are more similar to spring-fed streams than during the rest of the year.

Groundwater-fed streams in alpine areas have been found to be more productive than other Alpine streams [68, 69], and are expected to be important refugia for overwintering aquatic fauna (Ward et al. 1999). Irons et al. [70] suggested that in order to survive the harsh northern winters, benthic macroinvertebrates move actively into unfrozen flowing waters. Conversely, organisms may actively move or passively be transported into glacier-fed streams and develop high densities under favourable conditions. This process was shown in Rotmoosache by the occasional occurrence of cold-stenothermic and rheophilic chironomids together with more eurythermic taxa, which usually occur only in lower reaches. Seasonal changes in temperature and other abiotic factors may, therefore, be an important prerequisite for larval development in glacial streams. As the temporal pattern of these shifts are likely to change from year to year (depending on weather conditions), high plasticity in aquatic invertebrates in terms of life histories (e.g. [71]) is necessary to enable relatively high levels of diversity and productivity to be sustained in glacial systems.

3.4 Benthic Macroinvertebrates and Emergence

As in the studies of benthic community structure in glacier-fed Rotmoosache and spring-fed Königsbach [57, 60, 72], there are distinct differences in emergence when the two different stream types are compared. Emergence numbers as well as emergence rate were much lower in glacier-fed streams than in spring-fed streams. The resulting converse pattern with altitudinal gradient, i.e. increasing emergence number and rate in the glacier-fed Rotmoosache but decreasing emergence in Königsbach, demonstrated the effect of different stream conditions. Severe abiotic conditions in headwater reaches of glacial streams exclude many taxa and result in comparatively low densities and emergence rates. At a certain distance from the glacier snout and with a relatively low stream gradient, more taxa and greater emergence numbers and rates are found. This pattern corresponds well with results of various studies [73] and to the general literature about alpine river zonation [59, 65, 73, 74].

The opposite pattern in the spring-fed Königsbach may be explained by increasing steepness with decreasing altitude, allowing the upper lower gradient stations to have earlier snowmelt from increased solar radiation and warmer temperatures. Although alpine springs can show considerable variation in the invertebrate fauna [75], spring-fed streams sometimes have higher zoobenthic densities and more taxa than kryal and rhithral habitats due to relatively benign and stable environmental conditions [65]. Species richness and composition in Rotmoosache and Königsbach compare well to the taxonomic composition of common aquatic insect groups in alpine and arctic streams. Several recent studies have provided data on aquatic insect assemblages in these kind of systems from various biogeographic regions [73]. The dominance of Diamesinae in glacial streams is common and contrasts with the dominant subfamily Orthocladiinae in spring-fed streams.

3.5 Emergence Windows

Several studies gave rise to a general view of seasonality in aquatic insect emergence. Cold-adapted insect taxa such as Diamesinae tend to dominate emergence in winter, Orthocladiinae in spring and autumn, and Chironominae and Tanypodinae in summer [76–79]. In Alpine streams at higher elevations, seasonal patterns are not that obvious. In the above study, most taxa emerged during the snow-free period with a maximum in summer. This pattern was held for the glacial stream where emergence reached maximum values in summer with maximum discharge, increased suspended solids, high flow variability, and substrate movement. The hypothesis that emergence activity should be lower in the 1–2 month period of increased glacier ablation could not be verified. There are several possibilities explaining this pattern. Moderate conditions during morning before the increased discharge around noon and maximum in late afternoon might enhance emergence at this time of the day, even in periods of seasonal floods and harsh environmental conditions. For certain tribes of Chironomidae, specific diel emergence patterns have been found [80, 81], with an evening and morning emergence pulse evident for several taxa.

Another causal relationship with emergence may be water temperature. Temperature might be the main cause for this seasonal pattern in emergence in two ways. First, many studies have shown that temporal cues for emergence are likely to be temperature related, with the onset of emergence associated with rising temperatures. At the same time, long winter ice cover has a profound effect on water temperatures. Water temperature remains relatively low or even near freezing for a long period, giving only several weeks of warmer temperatures for enhanced larval growth. Several studies showed that latitude and altitude had a strong influence on emergence period, resulting in synchronous emergence patterns by aquatic insects [76, 82, 83].

Aquatic insects must overcome the harsh conditions of their aquatic habitat in the same manner as insects exposed to harsh conditions on land. By possessing various morphological structures and physiological traits, most stream insects are adapted to the dynamic and cold aquatic habitat of alpine landscapes. On land, heavy winds, cold temperature, rain, and even snow in summer can prevent or decrease reproductive success. Most likely, emergence during the warmer summer months when the probability of experiencing favourable climate conditions on land is higher than for the rest of the year was an evolutionary advantage for many Alpine stream insects. The combination of declining snow cover, rising water temperatures, and warmer and more stable conditions on land probably promoted aquatic insect emergence in summer. Alpine stream insects, in glacial streams in particular, must be adapted to meet a variety of environmental conditions in the water and on land.

Abiotic conditions during winter in the Rotmoosache were different than summer. Water temperature and discharge were low but stable and mainly groundwater-influenced, channel stability was high and snow cover inhibited solar penetration. These conditions were similar to those found in other winter studies of Alpine streams [69]. In this and other year-round studies, changes in species assemblages reflected observed environmental changes with additional taxa appearing in winter and some typical summer taxa disappearing [60, 61, 72, 84]. Increases in biomass and taxon richness of macroinvertebrates in winter have also been observed [84, 85]. The benthic fauna of the Rotmoosache changed in composition between summer and winter, and significant differences in Chironomidae abundance and taxon richness were found [57].

3.6 Abundance and Taxon Richness

In the above study, the total abundance of macroinvertebrates was mainly influenced by a single taxon such as *B. alpinus* or *Diamesa* spp. that can produce extremely high numbers of larvae when conditions are favourable. Results showed that abundances can be high or low in streams covered by ice and snow, varying by month and year. Although there were major differences in *B. alpinus* density, there were no significant differences in abiotic stream conditions between the two study winters. In contrast, autumn conditions differed between the two study years. The favoured emergence period of *B. alpinus* in north Tyrol, Austria is from mid-July to early October [86]. Autumn in 1996, typified by low discharge and a stable stream bed, lasted from the end of August to the end of October and provided good conditions for oviposition and egg hatching. In 1997, in contrast, high discharge lasted until mid September, thus this benign environmental period was much shorter. Eggs and newly hatched *Baetis* larvae may have been killed or carried downstream by high flows in 1997, and as a result *Baetis* density in the stream in winter was much lower than the year before. The duration of stable autumn conditions are probably important for other aquatic insects in these streams as well.

3.7 Species Composition

Seasonality is a common feature of macroinvertebrate assemblages in glacial streams in the Swiss Alps [84]. In the above study, assemblages also separated according to their composition, which varied seasonally. *Diamesa* spp. are characteristic and abundant in glacial streams during summer [59, 65, 87]. In the Rotmoosache, Diamesinae densities were high in summer, but only a few individuals were found in streams covered by snow. High emergence rates for *Diamesa* spp. in the Rotmoosache occurred in August in 1997 [58], corresponding to the time when their abundance in the benthos decreased. As early instar *Diamesa* larvae were in low abundance during winter, this taxon may overwinter as eggs or as larvae in the hyporheic zone.

The mayflies, *Baetis* and *Rhithrogena*, are common genera of the fauna in glacial streams some distance from the glacier snout [65, 88]. In the Rotmoosache, numbers in summer did not give a realistic impression of the densities these taxa can reach. For *B. alpinus,* nearly 70% of all individuals were found in winter, for *R. loyolaea* ~66% and for *R. nivata* Eaton around 85%. Thus, the contribution of these taxa to the total insect production in glacial streams is likely underestimated when extrapolated solely from summer sampling. *Capnia vidua* is typical of Alpine streams and emerges in early spring. Previously this species had not been described for glacial streams at altitudes >2,000 m asl, possibly because they were only collected in summer when early non-identifiable instars occur. Adults of *C. vidua* were collected from April to May 1997 in emergence traps in Rotmoosache [58], which corresponds to the collection of larvae of the same cohort from November to May. Most other Plecoptera (*Rhabdiopteryx alpina*, *Protonemoura* sp., *Leuctra alpina* Kühtreiber) also typically emerge in spring and thus must develop during winter. Many Plecoptera, so-called winter stoneflies, can develop at low temperatures in streams during winter [89].

The low presence of predators as a typical feature of glacial streams [65] is not completely true for the Rotmoosache. Several predators were found, mainly during autumn and winter, including *Clinocera* sp., *Dicranota* sp., *Dictyogenus fontium* (Ris), *Perlodes* sp., and Hydracarina. Such predators may influence benthic community structure during parts of the year when predator abundances are high. Crustacea and Nematoda prefer the hyporheic zone, and as groundwater prevails in autumn and winter, their contribution to the benthic fauna increases. Malard et al. [90] found that Oligochaeta distribution in a glacial stream was closely related to interactions between surface water and groundwater. They concluded that the hyporheic zone can be a colonization source for benthic habitats in periods of suitable environmental conditions. Crustacea and Nematoda can be indicators of the active seasonal migration of species between hyporheic and benthic habitats.

3.8 Life-Cycle Strategies

The survival of populations in the extreme environment of glacial streams obviously requires special strategies. Organisms living in high Alpine streams have developed physiological and behavioural adaptations to cope with low water temperature and freezing in winter (reviewed in [74]). Two evolutionarily successful strategies in glacial streams are adaptation to unstable stream conditions during summer (summer species) or avoidance of these conditions (winter species). Both strategies were found in the Rotmoosache: *Diamesa* spp. occurred as a summer species and most Plecoptera and Ephemeroptera as winter species. Periods of relatively stable conditions in glacial streams are early summer (between snowmelt and icemelt), autumn and winter. The early summer window is short [60], while autumn and winter provide a longer period of stable conditions. The winter months also have disadvantages: water temperature is always low, near 0°C, streams are usually snow-covered, and autotrophic production is low. A snow cover of 50 cm attenuates 99.5% of the photosynthetically active radiation [27], and summer species can overwinter in diapause as larvae or eggs. Winter species must develop in darkness and at low temperatures, which slows growth and prolongs development time [89, 91], although better food quality and stable stream beds may compensate for the disadvantage of low temperature.

4 Fishes of Alpine Waters

Alpine waters are sensitive ecosystems with unique features and resources. Extreme environmental conditions (altitude, gradient, low nutrients, duration of snow cover) shape special habitats that are only suitable for highly-adapted fishes. Only cold stenothermic species can inhabit Alpine waters. During spawning and the period of egg development, water temperature is low and can reach 0°C. Therefore, only a few of the native fishes were able to colonize and inhabit Alpine waters. In the last decades, non-native cold water resistant fish appeared in many Alpine waters. Non-native species have inhabited alpine lakes since the late 19th century (*Salvelinus namaycush* were stocked in 1886 in small alpine lakes in the Swiss Alps [92]) and started to reproduce in many lakes. Over the last decades in many Alpine streams, non-native, cold stenothermic species have established self-reproducing populations and appear well-adapted to the harsh environmental conditions.

Because of public interest and their role as ecological indicators, the distribution of fishes in Alpine waters is well known. On the other hand, knowledge is limited regarding the limits of occurrence, persistence, and long-term survival of fish populations in Alpine waters. Important characteristics and potential limitations for fishes in Alpine waters still need to be identified. Alpine ecosystems are threatened by human activities. Fishes are particularly affected by hydropower use, but also by habitat area, intensive agriculture, and last but not least by management activities including poor fish management.

4.1 Fish Communities in Natural and Artificial Waters

4.1.1 Lakes and Reservoirs

There are a large number of small lakes in the Alpine region, along with many reservoirs used for hydropower that were built after 1950. In Switzerland, there are currently 156 large dams. Without human influence, most of these lakes and reservoirs would be fishless, but due to management and stocking activities, most Alpine lakes are now inhabited by fishes. High Alpine lakes (>1,800 m asl) are primarily inhabited by the native brown trout, and to a much lesser extent by alpine char. Other species include the minnow, bullhead and exotic lake trout (*Salvelinus namaycush*). In lakes of the subalpine region (1,000–1,800 m asl), the alpine char is more common, as are brown trout, minnow and bullhead. Brown trout reproduce mainly in rivers that flow into and out of lakes, although natural reproduction can also occur in the lakes themselves. Brown trout must compete for resources with *Salvelinus namaycush* and brook trout (*Salvelinus fontinalis*), which reproduce successfully in many Alpine lakes, primarily in Canton Ticino. In addition to *Salvelinus namaycus* and brook trout, rainbow trout have also been stocked in alpine lakes. Intensive fish management in alpine lakes often causes problems with conservation efforts aimed at amphibians.

Fig. 3 Alpine brown trout from the River Vorderrhein, Switzerland. The brown trout is the most important fish species in the alpine region and is stocked in many streams for recreational fishery and trout conservation and population management

4.1.2 Springs and Streams

The brown trout (*Salmo trutta fario* L.) is the indigenous trout species in Europe and is the most important fish in Alpine running waters (Fig. 3). Brown trout inhabit springs, streams and rivers, and also lakes. There exists a long history of artificial stocking of brown trout in Alpine waters. Good quality as a sport fish and the flavourful meat have encouraged fishermen to introduce brown trout into waters that were devoid of fish. Brown trout have established themselves in many streams as a breeding stock and persisted without additional stocking. Their life cycle in Alpine waters is similar to lowland streams, but the different life stages must be adapted to the harsh alpine conditions. Here, eggs undergo a critical period of early development before they reach the larval feeding stage. The temperature during incubation (period when the egg is developing in the gravel bed) can reach low temperature, even 0°C. During the incubation period, Friedl [93] observed low mean temperatures of 0.65°C in the Julia River (Canton Grisons, Switzerland), where the eggs only needed 140 degree-days from fertilization to hatching (Fig. 4). In lowland populations, about 400–460 degree-days are needed [94]. In order to avoid scouring of the stream bed, the emergence from gravel must occur before bedload transport processes become active from snowmelt in spring. Because of the slow growth of alpine trout populations, sexual maturity is reached late [93]: males at an age of 3+ (>50% are mature) and females at 4+ (>50% are mature).

Brown trout populations can reach considerable abundance and biomass in Alpine streams. In 87 different alpine Swiss streams >1,500 m asl, an average abundance of 1,315 ind ha^{-1} with a biomass of 72 kg ha^{-1} was observed (A. Peter, unpublished data). Major limiting factors for trout are altitude and slope. Typically, streams at altitudes above 2,300 m asl are rarely inhabited by

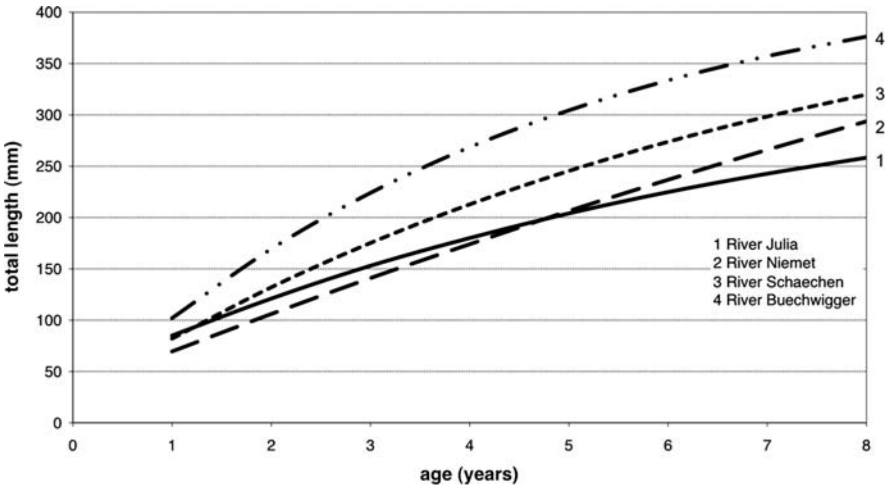

Fig. 4 Growth curve of brown trout in Swiss Alpine rivers

trout. The highest site of occurrence in the Swiss Alps for brown trout is in a lake at 2,823 m asl [95]. No clear limit is known for channel slope, although slopes <50° do not limit the occurrence of trout but slopes >150° impede natural reproduction [96].

4.2 Occurrence of Fish Species above 1,800 m asl

Timberline in the Alps is at approximately 1,800 m asl, depending on local climatic conditions. Above this elevation, the following native fish species still occur in the Swiss Alps [95]: brown trout (*Salmo trutta fario* and *Salmo trutta lacustris*), grayling (*Thymallus thymallus*), char (*Salvelinus alpinus*), bullhead (*Cottus gobio*), minnow (*Phoxinus phoxinus*), stone loach (*Barbatula barbatula*), rudd (*Scardinius erythrophtalamus*), burbot (*Lota lota*), and perch (*Perca fluviatilis*). The three exotic fish species also have reproducing populations in the Alps with established populations above 1,800 m asl: rainbow trout (*Oncorhynchus mykiss*), brook trout (*Salvelinus fontinalis*), and lake trout (*Salvelinus namaycush*). In order to increase sport fishing in some lakes and other closed waterbodies, stocking still occurs for these exotic species. In Italy, high fishing pressure is typical and has resulted in using strict management and massive restocking programmes [97].

4.3 Lake Inlets and Outlets

Lake inlets and outlets are special ecosystems because of their nearness and direct connectivity to the lake. In the Engadine, Graubunden, Switzerland, the outlet

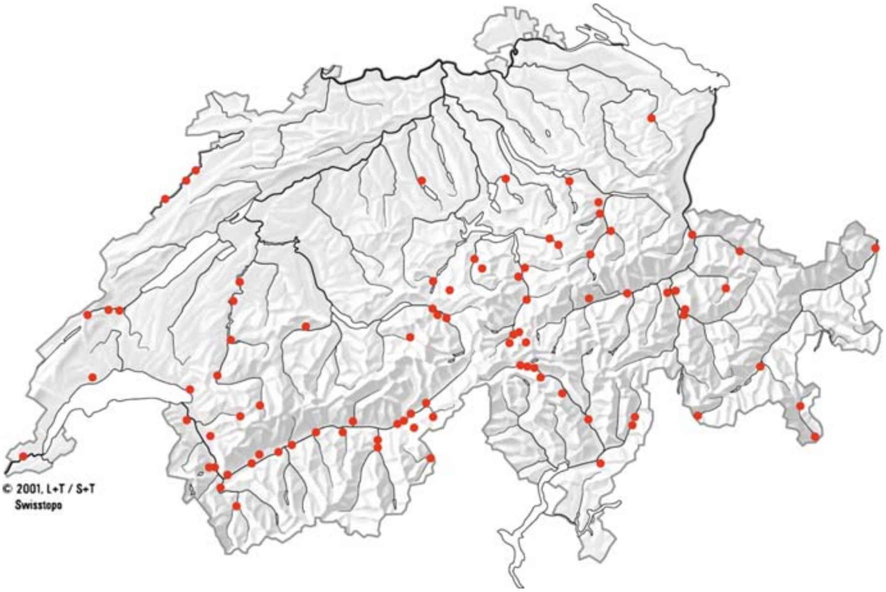

Fig. 5 Map of rivers with hydropeaking in Switzerland

stream of Lake St. Moritz (elevation 1,768 m asl) has a self-reproducing grayling population. This population is considered to be the highest elevation population in Europe. Lake inlets often have special importance for the reproduction of fish populations. Resident brown trout of mountain lakes depend on suitable spawning sites in streams, and often the inlet stream forms a small delta that provides suitable spawning habitat for brown trout and minnows.

4.4 Flow Regulation and Hydropeaking

Alpine streams and rivers are extensively used for hydropower production (Fig. 5). In Switzerland, 156 large dams and >1,600 hydropower plants exist, most situated in the Alpine region. Around 58% of Swiss electrical production originates from hydropower, and in Austria this amount is even higher at 68% [98]. In Italy, hydropower production is only 20% [97] of the total energy production. In all three countries, the largest proportion of hydropower production occurs in the Alpine region. The impacts on fish populations are many and caused by different components of hydropower use such as effects from dams and reservoirs, residual flow, hydropeaking, barriers affecting upstream and downstream migration, and flushing sediments from reservoirs and water.

Flow regulation mainly concerns residual flow and hydropeaking. Residual flow reaches are affected by the lack of natural flow patterns and by decreased water

Fig. 6 Release of water into the River Vorderrhein, causing a hydropeaking regime

depth and flow velocity. In addition, fine material accumulates to a higher degree in the stream bed and a general tendency for clogging is observed. Experimental flood programs carried out in a minimum flow reach of the Spöl River increased the fisheries potential [99]. As a consequence of the experimental floods, the quality of fish habitat and spawning areas improved. Formerly clogged riffle habitats were cleaned by the experimental floods.

There is no single population parameter for brown trout suited to characterize the effects of minimum flow, and no data suggest the potential of a suitable minimum flow [100]. The effects of reduced flows vary with stream structure, hydro-physical variables, and fish species. As such, trout biomass and density can increase or decrease in minimum flow reaches. If the minimum flow discharge is not suitable to maintain fish habitats, a sharp decrease in trout density and a disruption in natural reproduction becomes apparent. Small-sized fishes such as bullhead generally do not respond to a decrease in water depth caused by minimum flows, whereas adult brown trout are clearly affected by a reduction in pool habitat.

Energy production from alpine reservoirs leads to rapid and daily changes in flow. Waters mainly affected by hydropeaking are medium to large rivers in valleys of the Alps and Pre-Alps [101] (Fig. 6). In Switzerland, every fourth river is affected by water releases below dams [102]. In most European countries there is no legislation regulating the management of hydropeaking. In a mountain stream

(River Oriège in the French Pyrenees), Lagarrigue et al. [103] observed a negative impact from hydropeaking on juvenile trout and subsequently their density and biomass decreased by 30%. No significant changes were reported for adults. Saltveit et al. [104] demonstrated with field experiments that sudden reductions in river flow can cause high mortality of juvenile salmonids through stranding. In the Alpine Rhine River in Switzerland and Liechtenstein, it seems that hydropeaking and fines are primary factors limiting the natural reproduction of brown trout [105]. As a management measure against the negative impacts of hydropeaking, Halleraker et al. [106] recommended dewatering in darkness (night) and using slow ramping rates ($<$10 cm h^{-1}). In theory, a variety of river engineering and operational measures can be used as mitigation to hydropeaking [101]. In a few rivers in Switzerland (e.g. Rivers Reuss and Linth) different measures have been used, although the outcome of these measures has yet to be documented.

4.5 Future Management Strategies

At high alpine conditions, brown trout and char form fragile populations because they live in marginal habitats. These populations should not be exposed artificially to competition pressure by stocking exotic cold-water adapted fish (*Salvelinus camaycush*, rainbow or brook trout). "Hot spot" areas such as stream reaches with low gradient, bifurcated stream beds and side channels are the main source of recruitment for brown trout and are highly limited ($<$1% of the total habitat area). To preserve or re-establish the ecological integrity of Alpine streams, essential source habitats should be identified and protected or restored. Alpine fishes should be considered populations that are highly specialized to high elevation environments and therefore require appropriate protection and conservation measures. Minimizing the impact of hydropower use on these fishes and assuring their natural reproduction will be a future challenge in Alpine stream management.

5 Perspectives

5.1 Habitat Template of Alpine Floodplains

The waters of alpine landscapes are highly complex and sensitive ecosystems. Glacier retreat has accelerated globally, increasing the probability that fundamental ecological changes will occur in alpine landscapes, in particular the ecology of running and standing waters. As the glacial influence diminishes over time, expected changes in water sources and hydrologic regime will certainly affect the distribution and abundance of organisms and the resulting functional characteristics of alpine floodplains. Presently, the characteristic features of glacial streams include: (a) an extended seasonal flow pulse controlled by temperature and radiation, (b) large diel flow pulses during summer, (c) low temperatures, (d) high turbidity during flow pulses, and (e) low channel stability mainly in recently deglacierized areas. Although glacial water is the primary source of water in most Alpine floodplains, groundwater sources also play a substantial role in the ecology of surface waters. Commonly, floodplain waters are fed by snowmelt in spring, glacial ice-melt in summer, a mix of subglacial and glacial water in autumn, and primarily groundwater in winter. Flow and temperature are highly coupled in glacial streams, and air temperature and radiation control both. The complex dynamics of surface and subsurface flow paths influence thermal patterns in Alpine streams. Temperatures decline rapidly in autumn from the decrease in air temperature and reduced solar radiation. The glacial influence on temperature decreases longitudinally because of heat exchange with the atmosphere and heat input by side-slope tributaries and spring brooks.

5.2 Flora and Fauna of Alpine Waters

The primary source of energy in alpine waters is autochthonous via benthic algae, and is a primary food resource for benthic invertebrates. Higher aquatic plants are essentially lacking, however mosses can be found especially in groundwater-fed streams. In general, algal biomass in glacial streams is high in spring before high runoff and late autumn, but extremely low in summer. Algae can also attain high levels in winter in stream reaches lacking ice cover, and groundwater streams usually have high algal biomass throughout the year. The complete algal diversity of Alpine waters is poorly known, with most focus on diatoms. Diatom communities are rich in taxa, among which are endangered or extremely rare taxa on the German Red List [10]. Alpine waters are refugia for these taxa, where they find suitable environmental conditions. The sensitivity of diatoms to environmental change makes them excellent for long-term assessment of ecological change in Alpine waters in response to climate change and anthropogenic stress. As such, it is imperative to maintain Alpine waters characterized with high ecological status as defined in the Water Framework Directive.

Zoobenthos of surface waters show substantial spatial and temporal variation. Spatial heterogeneity of surface waters may enhance the overall biodiversity of zoobenthos, and some streams may provide refugia during harsh periods in glacial streams. In general, aquatic insects comprise a substantial proportion of the zoobenthos in surface waters, with Chironomidae being most common. The abundance and diversity of surface zoobenthos typically increases downstream as stream environments become more physically favourable. The *Diamesa*-groups (chironomids) usually dominate upstream glacial streams, with Ephemeroptera (*Baetis alpinus* and *Rhithrogena* spp.), Plecoptera, and Trichoptera becoming more common in lower sections. Groundwater streams also host the latter groups, at times reaching substantial numbers. Lake outlets provide another habitat type that often is inhabited by a distinct assemblage of zoobenthos.

Fishes in Alpine floodplains are limited to cold stenothermic species such as the brown trout, and many alpine lakes are currently stocked to sustain the fishery. Water abstraction and flow regulation severely constrain the management of the fishery in Alpine waters today. The effects of climate change on the fishery are difficult to predict but could facilitate the upward migration of more cool water fishes in the future. The implications of these new fishes on aquatic food webs are not certain but could be substantial.

5.3 Future Considerations

During the last 10,000 years, glaciers in the Alps have advanced and receded at various times, but glacierized areas have decreased rapidly in the last 20 years. A snowline increase and loss of ice will markedly influence the flow regime and sediment dynamics of Alpine streams. Proglacial areas contain large amounts of unconsolidated sediment susceptible to fluvial transport. Snowmelt will have a greater influence on the shape of the hydrograph than today; annual discharge may decline and rain-induced floods will occur more frequently, especially in summer. An expected consequence of glacier recession is that kryal reaches will move upstream and temperature will increase in surface-connected channels. At a longer time scale (e.g. 100 years), most Alpine waters will likely shift from glacier-dominated to snow-dominated ecosystems. An increase in the presence of intermittent and groundwater (snowmelt fed) streams is highly plausible.

Although Alpine streams act as integrators and centres of organization within the landscape, the ecological effects of landscape change on Alpine streams are difficult to predict. Low channel stability could result in reduced abundances of benthic macroinvertebrates, and increased water temperatures could lead to extinction of cold water species and colonization by new species from lower elevations. Glacial retreat may fragment populations of aquatic invertebrates and fishes with variable affects on population persistence and genetic structure. Glacier retreat may promote the enlargement or formation of proglacial lakes, the outlets of which are characteristically stable habitats for zoobenthos. The insularity of high mountain areas will increase and freshwater habitats probably become more fragmented or temporary. Previous kryal systems will follow expected successional changes in habitat properties with concomitant changes in biodiversity and ecosystem function. As glacial retreat continues upslope, the mosaic of successional stages will change longitudinally and laterally. Longitudinally, new or younger streams will be found at higher elevations and will exhibit a long-term colonization pattern. Colonization of these new streams will be constrained by biogeographic and local factors such as distance, elevation, channel stability, and stream temperature, and biotic factors such as species-specific life histories and dispersal behaviours.

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Part IV
Case Studies

Integrated Environmental Management of Hydropower Operation Under Conditions of Market Liberalization

Bernhard Truffer

Abstract Electricity generation from hydropower is responsible for a large part of the human impacts on water resources in the Alps. Attempts at finding an encompassing mitigation approach to these impacts are often confronted with high levels of societal conflict. Therefore, an integrative approach has to consider both ecological and political/social aspects of this economic activity. In the following, we will present an integrative mitigation approach that was developed as an eco-label for electricity in Switzerland. This label was developed and implemented in the early 2000s and has meanwhile been accepted as an environmental standard for hydropower operation in many other countries. The chapter emphasizes the interdisciplinary competencies needed as well as the potential benefits of an integrative approach to the problem.

Keywords Eco-labeling, Hydropower, Integrated concepts, Interdisciplinarity, Switzerland

Contents

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1 Conditions for the Development of an Integrated Approach

Hydropower use for electricity generation is responsible for a wide array of environmental disturbances to river systems. Over the past decades, aquatic science research has been successful in identifying a considerable number of relationships that exist between plant operation and ecosystem quality. This increase in scientific knowledge was, however, not matched by a corresponding reduction in environmental impacts stemming from hydropower. One of the major reasons for this situation is that political, economic, and social aspects are neglected in purely scientific assessments of environmental impacts.

The situation may be illustrated well by the development of the public discourse on hydropower in Switzerland over the past few years. As 60% of Swiss electricity generation stems from hydropower, its environmental impacts have time and again raised political debates over the past 50 years. More recently, the liberalization of electricity markets has opened up new windows of opportunity for the development of new approaches of mitigation. One example of an integrative concept was developed by an interdisciplinary team of researchers in the late 1990s. It has been implemented in the currently operating eco-label for sustainably produced electricity in Switzerland (naturemade star).

We will present this procedure by first reviewing the state of aquatic science research on alpine river systems. The limited implementation of this knowledge in Switzerland will then be explained by a “social dilemma” among different stakeholder groups. In order to overcome this dilemma, a coupled research and negotiation process was initiated by the research team with the aim to establish a green electricity label for hydropower. By reconstructing the development process of this label, we claim that an integrated approach of environmental and social aspects of water use activities is necessary in order to effectively mitigate the associated impacts.

2 Understanding the Environmental Impact of Hydropower Operation

Reservoirs and dams are responsible for a wide variety of environmental problems [1]. According to a review by Gleick [2], almost 500,000 km^2 of land are inundated worldwide by reservoirs. Sixty percent of the 227 largest rivers on earth are strongly or moderately fragmented by dams and diversions [3]. Most of the dams in mountain regions are used for hydropower production. Switzerland, Austria, Norway, and Japan have the highest hydroelectricity production per surface worldwide [4]. This high density of power plants may have considerable impact on biodiversity and ecological stability of mountain ecosystems: In the European Alps, 79% of the river reaches are influenced by hydropower operations [5].

At the local level, hydropower construction and operation is associated with a number of serious environmental problems: water diversion, interruption of fish migration, hydropeaking, reservoir flushing and inundation of landscapes, and alterations in bio-geochemical cycling [6]. This assessment led early on to dedication of a considerable amount of research activity to the ecological impact of dams [7]. Different research agendas have been developed in recent years to understand and assess the functioning of natural and modified aquatic systems and to predict their reaction to river restoration projects. The integrative concept of "river health" has been proposed to provide a conceptual framework for these analyses. However, it is not an easy task to develop an operational protocol for the assessment of "river health." Such efforts have been translated recently into monitoring programs run by political authorities such as the European Union Water Framework Directive or the Swiss Modular Concept for River Assessment [8].

Despite the massive efforts expended to develop concepts for river assessment and to determine the negative effects of damming and hydropower operation on aquatic ecosystems in the Alps, very few success stories can be found regarding the actual mitigation of these impacts. This failure of linking scientific results with actual problem resolution was not due to a lack of public awareness. Quite the contrary is true, in particular for Switzerland.

3 Understanding the Sociopolitical Context of Hydropower

The mitigation of hydropower impacts on Alpine ecosystems has been a topic of considerable public debate in Switzerland (see [9]).

3.1 The Political History of Hydropower in Switzerland

The public discourse about hydropower in Switzerland went through a number of quite clearly demarcated phases. In the first phase, which started between 1880 and 1914, the electrical utopia based on "white coal" sparked a national consensus to develop alpine hydropower plants [10]. However, due to technical limitations, the majority of currently running large-scale hydropower plants in the Swiss Alps was constructed in the 1950s and 1960s. As a result of this rapid expansion, almost all major streams in the Alpine region are now impacted by hydropower plants and their operation [6].

Because of the increasing visibility of negative impacts on ecosystems, landscapes and local communities, public opinion began to turn against plans to invest in new hydropower plants in the late 1970s and early 1980s. A number of environmental grass roots movements fighting against hydropower gained widespread public attention. This change in public perception culminated in a fierce political struggle over the renewal of the Swiss Water Protection Law at the end of the 1980s that fixed, among other subjects, minimal flow requirements for Alpine streams impacted by reservoirs.

The renewal of the Water Protection Law led to a strong political opposition between several parties: Environmental organizations struggled for the protection of the last untouched river stretches in the Alpine mountain valleys while electric utilities lobbied against "unproductive" water running down the river. Given the high share of public ownership in hydropower plants, interest conflicts also extended towards federal authorities, supporting the Water Protection Law, and regional (cantonal) authorities in their role as shareholders and tax receivers of hydropower plants.

After the law passed, the political debate about the sustainable management of hydropower came to a standstill. The law could only be applied in the context of a renewal of the water use licenses. In Switzerland these licenses typically run over 80 years and therefore no major changes in operation could be expected before the year 2020. Furthermore, the deregulation of electricity markets begun to shape expectations in Switzerland. This new market order put pressure on hydropower operators to reduce cost and to act as competitive firms. Environmental requirements were considered as a direct threat to economic survival of the plants.

3.2 Green Power Products as an Incentive for Sustainable Hydropower Operation

Nevertheless, the new market order also opened up opportunities for utilities to deal with environmental problems. Experience from other countries with deregulated electricity markets showed that consumers are willing to pay extra for electricity with low environmental impact. A number of incumbent electric utilities, as well as newly emerging green power marketers, began developing green power products to differentiate themselves from other suppliers [11].

A number of studies identified key conditions for success (see [12, 13]) for utilities wanting to operate in these markets. In particular, it was found that the willingness for surplus-payments was tied to the expectations that green products should be directly tied to environmental mitigation measures [14]. This criterion has specific implications for hydropower compared to other renewable energy sources [15]: there are already considerable production capacities at competitive prices all over Europe. More importantly, building new power plants is often associated with additional impact on aquatic ecosystems. Therefore, the environmental benefit associated with buying green electricity from hydropower could be negative if green electricity sales led to the construction of many new plants. Environmental benefit would be better if the eco-label created an incentive for existing power plants to adopt less environmentally disrupting operation modes. However, no widely shared standard for sustainable hydropower operation existed up until the late 1990s internationally [15].

3.3 The Green Hydropower Label as a Means to Overcome a Social Dilemma

Given this political history of hydropower in Switzerland, a standard for sustainable hydropower operation was likely to provoke major political reactions. However, expectations associated with market liberalization also partly motivated actors to reconsider their original interest positions. However, solutions could not be found because the interlocked nature of the decision problem represented a classical "social dilemma" [11]. A social dilemma is present if decisions of two actors depend on each other and if both actors are forced to select a sub-optimal strategy of conduct to minimize their potential losses. An optimal solution would only be realized if each party could trust the other.

In the Swiss hydropower "game" either party could not trust the other (see [9] for more detail). For environmental NGOs, for instance, it became clear that with market liberalization new approaches for regulating the environmental impacts of the electricity sector had to be found. Hydropower was perceived as being much less destructive than most of the low-cost fossil fuel alternatives. For hydropower operators, market opening was associated with increasing pressure on the cost side. Differentiation along environmental criteria seemed at least one promising way to improve their income especially if green consumers were prepared to cover additional costs associated with environmental upgrading measures [14].

4 The Development of a Swiss Green Hydropower Standard

Having understood the dilemma structure of the situation, the preconditions for promoting less damaging ways of hydropower operation could be formulated much more precisely. A team of researchers at the Swiss Federal Institute for Environmental Science and Technology (EAWAG) started a "Green Hydropower Project" in 1997 [16]. It identified the problem field of green electricity as an interesting research opportunity because a whole series of ecological and economical research questions could be addressed by it. Furthermore, the development of an eco-label for Green Hydropower promised to have a high impact on the mitigation of impacts on Alpine ecosystems. The interdisciplinary cooperation within a clearly defined project structure and the careful management of external relationships with stakeholders led ultimately to the achievement of all the initially set goals [17].

The development of a concrete certification procedure had to avoid the pitfalls of both established aquatic science assessment protocols and existing hydropower eco-labels. Protocols are in general very extensive and detailed but focus on assessing "deficits." Often it is a long way to go from there to proposing actual mitigation measures. This means that existing protocols are mostly scientifically adequate but not easily applicable in practice. The prevailing eco-labels for electricity of the late 1990s, on the other hand, used short-hand criteria (such as

Table 1 Structure of EAWAG assessment procedure for Green Hydropower (see [18])

Management fields / Environmental fields	Minimum flow	Hydro-peaking	Reservoir management	Bedload regime	Power plant structures	Individually adapted restoration measures
Hydrological character						
Connectivity of river system						Eco-investments
Sediments and morphology		Basic requirements				
Landscape and biotopes						
Biocenoses						

size or age) for determining environmentally preferable hydropower plants. These criteria, however, have virtually no correlation with environmental impact. Therefore, they were easy to apply but essentially meaningless.

The newly developed Green Hydropower criteria in turn related to the mitigation of local impacts from hydropower plants while putting them into an operational context. As a first step, "basic requirements" were defined along two dimensions (see Table 1):

- The first vector lists the relevant ecological domains of the river system: the hydrological character; the longitudinal, vertical and lateral connectivity; aspects of sediment transport and morphology; landscape features and biotopes as well as relevant aspects of aquatic ecosystems.
- The second vector outlines the action fields for hydropower operators: regulations for an environmentally compatible minimum flow regime; measures to mitigate negative effects of hydropeaking; management advice for reservoir and sediment management; and conventions on how to design power plant structures like channels, fish passes, or turbine inlets.

Standards were formulated for each of the 25 fields of the matrix by defining environmental goals, specific criteria as well as methodological support information [18]. The matrix approach helped in structuring the problem area and converting ecological know-how (the rows of the matrix) into management goals for hydropower operation (the columns). However, the individual criteria were quite difficult to quantify for a general case [6].

As a consequence, basic requirements were complemented with the additional demand that a certain amount of money earned by green electricity sales should be invested into additional local upgrading measures. These so-called eco-investments were financed by a fixed mark-up on the price of a kilowatt-hour of the electricity sold to green customers. The measures to be implemented as eco-investments were determined in a participatory process encompassing local interest groups from the power plant, local environmental NGOs, and local governments. Proceeding in this way, the eco-label could guarantee a high mitigation level of impacts from the respective power plant and operators had a strong argument to prove that their electricity offer "makes an actual difference" to the environment.

5 Effectiveness of the Eco-Label for Mitigating Hydropower Impacts

The Green hydro assessment procedure was implemented in the official Swiss eco-label for green electricity naturemade star in the year 2000. Meanwhile, about 40 plants have been certified according to this procedure. These plants account for approximately 3% of electricity produced by hydropower in Switzerland. About 10.2 Mio SFr were collected for local eco-investments directly paid out of green electricity sales (communication by Swiss Association for Environmentally Sound Energy, VUE). Therefore, only a minority of plant operators decided to upgrade their operation with regard to sustainability standards. This somewhat sobering result does however not only apply to green hydropower products, but may be observed for green electricity markets, in general [19]. Reasons for this limited success are varied. Not least, they were due to the uncertain prospects of electricity market liberalization that have prevailed in Switzerland since the year 2000. Nevertheless, the implementation of such a broadly based, integrated assessment procedure also created additional benefits for the mitigation of environmental impacts from hydropower. For instance, certain utilities decided to use the standard as a basic orientation for upgrading their portfolio of plants. At a more political level, the proof of existence of a feasible, encompassing standard facilitated cooperative solutions in later debates on hydropower impact mitigation, for instance in the definition of suitable areas for constructing new power plants. Finally, the standard also earned considerable recognition outside Switzerland and helped to structure the debate also at an international level.

In order to achieve these results, two critical conditions for success may be identified: first the integration of different disciplinary knowledge stocks for developing a feasible product standard and second, the explicit consideration of facts and values in the process [17]. A good example of the latter is associated with the assessment of general environmental impacts from hydropower. This technology has an extraordinarily favorable record regarding greenhouse gas emissions and other life cycle characteristics. On the other hand, it can perform very poorly with respect to local impacts on ecosystems. However, there is no scientifically sound procedure, which could measure the objective trade-off between these two types of impact. An explicit solution has to be negotiated by the relevant interest groups. Another key issue concerned the quality level of the basic requirements. After a long negotiation process, the level was set similar to the ecological quality, which a newly licensed power plant would reach. This in itself proved to be a major breakthrough regarding the political conflict lines encountered at the start of the project.

Developing integrated approaches for the mitigation of environmental impacts from hydropower thus has proven to be a highly challenging task for environmental research. It has to take the best of environmental research into account which then has to be integrated into encompassing concepts identifying healthy ecosystems. Additionally, also the social context conditions have to be actively considered. Often this means that historical, political, and economic contexts have to be analyzed on an equal footing. Finally, a very explicit consideration of facts and values is necessary, if solutions are to be broadly shared by different interest groups. If these points are taken into account, however, the impact of environmental science knowledge is likely to be increased substantially.

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Downstream Relevance of Reservoir Management

Alfred Wüest

Abstract The management of dams serves many purposes and goals all over the globe, and has important consequences for the downstream rivers and lakes. Among the more than 50,000 so-called large dams, the biggest are located in alpine regions. As a result, the water residence time in heavily dammed alpine valleys typically increased from a few days to several weeks, hydrological regimes shifted seasonally and sediment transport often decreased to half of its natural value. The occurrence of high flows responsible for most particle transport is reduced and particles are trapped behind the dams. These changes modify particle concentrations and particle size distributions, thermal regimes and water quality in downstream waters. As a result, downstream rivers and pre-alpine lakes often experience significant alterations in particle, carbon and nutrient cycling. Also described are common mitigation measures that are often applied in newly-planned damming management.

Keywords Downstream effects, Hydro-peaking, River temperature, Sediment retention, Water abstraction

Contents

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1 Introduction

The damming of waterways and management of reservoirs are aimed at many different purposes and goals. Whereas in the European Alps the emphasis is on hydropower production (Fig. 1), there are many additional objectives for dams in other regions of the globe, such as irrigation (China, India), water supply (USA, UK) and flood control (China, USA). Reservoir management affects upstream waters – via processes in reservoirs and inflowing rivers-as well as downstream rivers, lakes, floodplains and estuaries (Table 1). In this article, the concept of dam management is not focussed on the optimization of reservoir operations, which is an important challenge in multipurpose dams with several simultaneous objectives. Here we review the effects on aquatic ecosystems as a function of reservoir management and operation. We start with the temporal structure of the modified flow as it affects sediment and heat transport. As a result, particle size distributions, thermal regimes and water quality in downstream waters are altered as well. The chapter closes with a short outline of mitigation possibilities and measures.

Fig. 1 The 250-m high wall is forming Lac Mauvoisin (211 Mio m^3) – a typical example of a reservoir in the European Alps – which contributes to the total storage capacity of about 1.3 km^3 in the Upper Rhone valley. The nominal power production for this reservoir is 350 MW. Picture: FMM SA

Table 1 Downstream effects of damming (some effects discussed in the text)

(1) Hydrology and hydraulics
- Changes of the seasonal hydrological regime and flooding frequency
- Change in total discharge (water loss by irrigation, diversion and evaporation)
- Short-term flow fluctuations (hydro-peaking)
- Water level fluctuations (draw-down zones in reservoirs and downstream lakes)
- Hydraulic changes in rivers and lakes
- Groundwater level shift (ex-, infiltration), salt (wedge) intrusions near estuaries
- Thermal stratification in reservoirs (“lake-type” versus “river-type”)
- Changes of temperature

(2) Sediment transport and river morphology
- Reservoir siltation and reservoir filling
- Reduction (and temporal shift) of particle transport, increased clarity/reduced turbidity
- Turbidity peaks (extreme values during reservoir flushing)
- Floodplain and bank deterioration (reduced flooding)
- Delta retreat and coastal erosion (reduced sediment supply)
- Geomorphological stagnation of rivers, riverbed erosion (reduced sediments and bedload), riverbed siltation and clogging

(3) Geochemical cycling
- Water quality changes in reservoirs (primary production, sedimentation, organic matter decomposition, oxygen consumption)
- Anaerobic mineralization processes in the sediment or bottom water alter water composition (e.g. by removal of nitrate, production of reduced substances such as Mn^{2+}, Fe^{2+}, $NH_4^+$and H_2S hazardous release of trace elements from sediments)
- Downstream water quality deterioration (altered nutrient composition, oxygen-depleted deep-water with reduced components)
- Nutrient trapping by sediment formation
- Salination by evaporation (irrigation)
- Production and emission of greenhouse gases (CH_4, N_2O)

(4) Aquatic and terrestrial ecology, fisheries
- Loss of terrestrial ecosystems
- Reduction of riverine/riparian/floodplain habitats
- New deltaic wetlands at reservoir inflows (waterfowl)
- Shift of species from “river-type” to “lake-type” in reservoirs
- Changes in all downstream aquatic systems (incl. wetlands, lakes, estuaries, sea deltas)
- Altered nutrient composition in downstream waters, (hyper)oligotrophication
- Disruption of connectivity, especially migrating fishes (such as salmonids, trout, sturgeon)
- Supersaturation (oxygen and nitrogen) by air entrainment at spillways and plunge pools
- Cold and warm water pollution

2 Hydrology Below Reservoirs

Damming leads to a seasonal modification in water flow. Different to the tropics where larger water surfaces increase evaporation, storage in alpine valleys (Fig. 1) hardly influences the water budget and the annual flow remains almost identical. In many heavily dammed alpine valleys, the storage capacity is typically 10–20% of the annual runoff. This manageable “life” volume can be laid over from summer (snow and glacial melting) to winter, the time of natural low flow and high energy

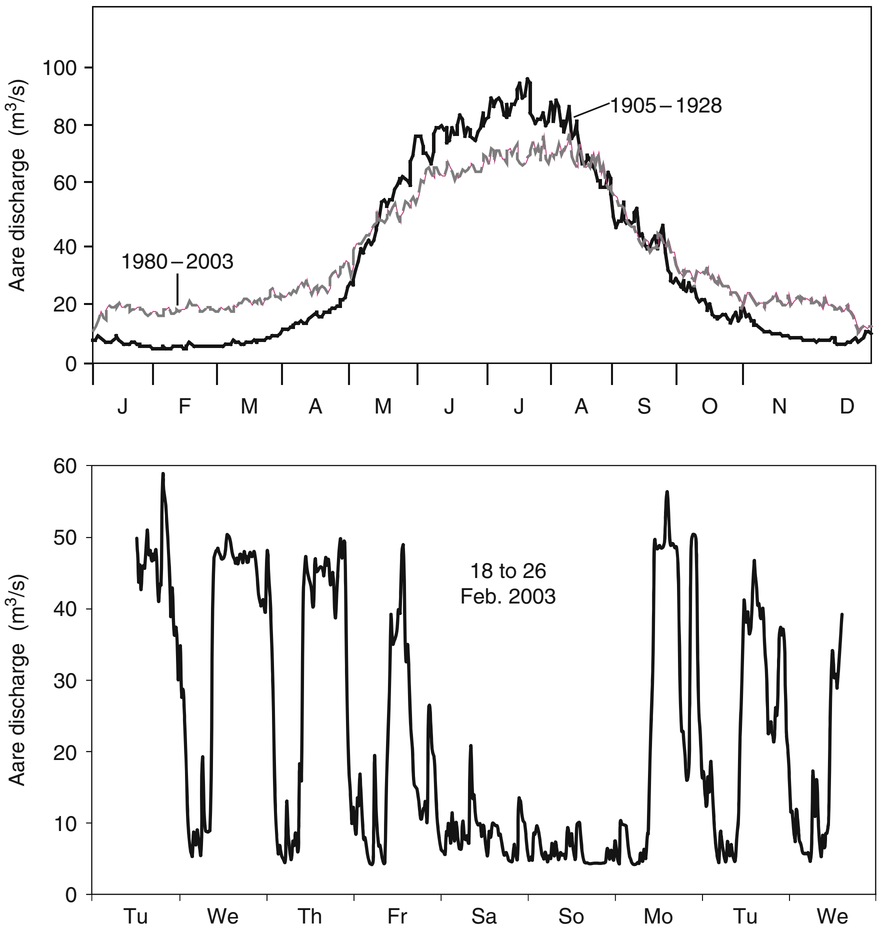

Fig. 2 There are two major hydrological modifications due to hydropower operation on River Aare: levelling of seasonal flow (*upper panel*) and hydro-peaking (*lower panel*). *Upper panel*: Seasonal flow (at Brienzwiler) before hydropower operation (1905–1928) and with a storage capacity of 200 Mio m^3 (1980–2003). Note the shift from summer to winter. *Lower panel*: Daily hydro-peaking fluctuations due to electricity production. Note low flow on weekends.
Data source: [16]; details in [17]

demand. An example is illustrated in Fig. 2a for the River Aare, which was dammed upstream of Lake Brienz during the twentieth century starting in 1928.

The consequences of an altered hydrology has been analyzed for the Upper Rhone River (upstream Lake Geneva) by Loizeau and Dominik [1]. The hydroelectric dams, constructed during the second half of the twentieth century, with a total storage of circa. 1.3 km^3 (circa. 20% of the annual flow to Lake Geneva), changed the characteristic flow regime of this river. The annual mean flow of 187 $m^3\ s^{-1}$ (with variations from 136 to 234 $m^3\ s^{-1}$; www.hydrodaten.admin.ch) from the 5,220-km^2 large catchment, has not changed noticeably. The flows have been altered threefold: (1) by a reduction of flow in summer, (2) by a drastic increase of winter discharge, and (3) by lowering the frequency and amplitude of extreme flows (flooding events as well as minimal flows). Before damming, daily discharges, exceeding 400 $m^3\ s^{-1}$, occurred on average 55 days per summer, whereas today such flows are observed only on 15 days [1]. Moreover, after the 1960s, larger floods (such as >600 $m^3\ s^{-1}$) disappeared almost completely, although the most extreme ones still occur. In contrast, during winter, minimal flows ranging between 50 and 60 $m^3\ s^{-1}$ before 1950 have been increased to 120 $m^3\ s^{-1}$ in the recent past. The overall effect of dams is a reduction in the frequency of extreme discharges at both, low and high flow ends (Fig. 2a).

The strongest effects on river flows, however, occur locally (1) in the abstraction (residual flow) sections between the water intakes and releases (where the river water is bypassed in pipes) and (2) in the river section immediately below the water outlets. In the first instance, environmental flow is an issue (Table 1), as diverting water can – in the extreme – completely empty river sections. The extent of water abstraction in alpine valleys is in some regions quite large, especially in central Europe, such as in the Swiss Alps as documented in Margot et al [2].

After power production, the released water is superimposed on the reduced discharge and causes unnatural flow characteristics in the river below the outlet: so-called hydro-peaking, an artificial sequence of low and high flow in a regular rhythm during workdays (Fig. 2b). As Fig. 3 illustrates, the hydro-peaking

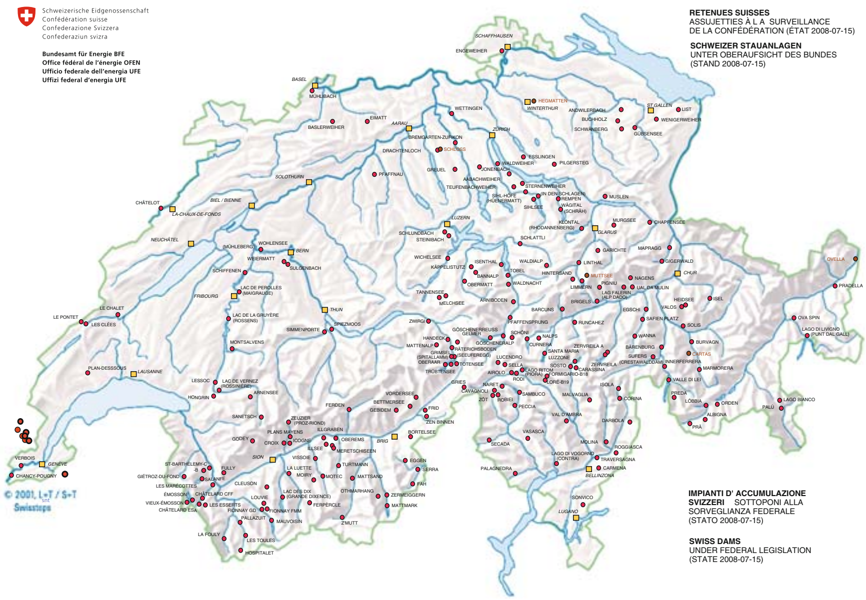

Fig. 3 Map showing the location of the 220 largest dams in Switzerland. The reservoirs have an integrated storage of more than 4 km^3 and provide a typical annual power production of 38 TWh. As the map indicates, basically all major alpine rivers are affected by abstraction or hydro-peaking

phenomenon extends over many alpine valleys**.** The ecological consequence of hydro-peaking is addressed in the chapter by A. Peter ([3], this volume).

Specific changes occur when former natural lakes are transformed into managed reservoirs, affecting not only downstream waters but also the former lake itself. As a result of damming and increasing the storage capacity, hydraulic changes become relevant: (1) water levels rise, (2) levelling of seasonal outflows, and (3) unnatural-deep (hypolimnetic) water release at the dam. Matzinger et al [4] used the Arrow Lakes in the Rocky Mountains as a case study to show that such hydraulic modifications can reduce a lake's biological productivity by up to 40%, primarily as a result of altered lake-internal flow paths that allow nutrients to creep through the reservoir without entering the productive surface zone. In lakes that act as a "nursery" for salmonid fry, such modifications may become critical [5].

3 Particle Trapping and Seasonal Particle Transport

Operating hydroelectric dams in alpine headwaters not only drastically cuts off high flow occurrences, but also reduces downstream sediment transport, as reservoirs trap particles and low competence streams carry lower loads. The intensity of sediment transport depends highly on the non-linear action of flow. Observations in alpine catchments reveal that often 80% of the annual transport occurs over short time scales [6] implying an enormous variability (triggered by precipitation) at sub-decadal scales. In a case study, Anselmetti et al [7] quantified the particle sedimentation in high-alpine hydropower reservoirs (constructed 70 years ago) on the Aare River (Fig. 2a). Mainly from the glaciated area of the catchment, circa. 270 kt yr^{-1} of glacial till and silt are swept into the reservoirs where only circa. 40 kt yr^{-1} remain in suspension and pass through, whereas circa. 85% of the input is deposited behind the dams. As an effect, only circa. one-third of the original natural downstream particle transport still reaches Lake Brienz today.

A comparable conclusion was found for the Rhone River from an analysis of the recorded suspended solids measurements in the twentieth century [1]. During the build-up of hydropower dams in the Upper Rhone Valley, the particle transport was reduced substantially. In Fig. 4, current seasonal particle transport is compared to the one measured a century ago in 1904/5. This data record allows a unique comparison that illustrates the drastic reduction of sediment transport during summer (Fig. 4b). Parallel to the water flow, particle transport during winter has increased (Fig. 4a) due to power production and the related drawdown of turbid upstream reservoirs. As a result the particle input to Lake Geneva today is about half compared to what it was under natural conditions before the 1960s [1]. The effect of damming also becomes evident when catchments of similar properties but different degrees of damming are compared. Figure 4c shows the reduced and shifted suspended solids transport of the River Aare (heavily dammed) in comparison to the River Lütschine (no dams).

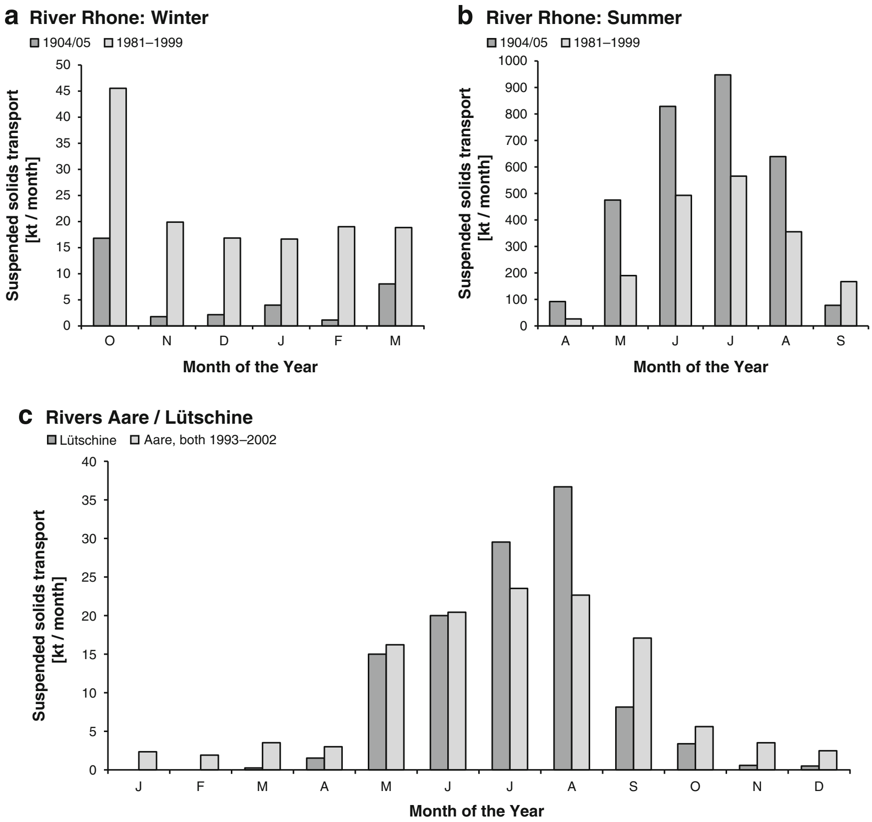

Fig. 4 **(a, b)** Particle transport in 1904/5 ([18]) before construction of the reservoirs in the Upper Rhone catchment, compared to recent (www.hydrodaten.admin.ch) particle transport. The unique, century-old dataset confirms the massive increase during winter (**a**) and massive reduction during summer (**b**). **(c)** Comparison of particle transport in two catchments with similar properties (see text), but distinctly different levels of damming. In the River Lütschine (no dams), the transport is concentrated to July/August and negligible in winter, whereas in the River Aare (200 Mio m^3 storage) the transport is reduced and significantly levelled over the season with a shift from summer to winter. Details in [17]

The level of reduction in suspended solids transport is quite typical also at the global scale. Using worldwide data, Vörösmarty et al [8] found alpine Europe among the regions with the strongest reduction in particle transport. In other regulated basins, particle transport is also often at circa. 50% of original natural levels. As an effect, the globally averaged sediment transport in rivers is reduced today by circa. 30%. Whether this reduction will increase with future regulation and damming of rivers depends also on the enhanced soil erosion, which largely compensates for the loss [9].

Dams not only reduce particle flux, they also shift it seasonally and fractionate the particle size composition, as only the fine fraction (< several μm, depending on the residence time) passes through reservoirs, while the mostly coarse particles are retained. Therefore, the particle size distribution today is shifted towards finer particles in downstream waters. A more detailed analysis would reveal that this relative distribution varies seasonally, as particle settling is a seasonal process, especially under ice cover in winter.

Pump-storage operations introduce even more complexity, as water is pumped/turbinated between reservoirs of different altitudes. As lower-elevation reservoirs (which are often natural lakes) usually contain less particles – as settling takes place along the flowpath – pumping homogenizes concentrations within the entire pump-storage system. The resulting higher turbidity in downstream waters often opposes the appealing economic advantage of steep alpine settings, ideally suited for pump-storage operations.

In remote alpine areas where anthropogenic pollution is limited, the retention of nutrients associated with trapped particles may become apparent. Case studies in two Rocky Mountain lakes (Kootenay and Arrows) indicated a reduced biological productivity, which is not compensated – as is usually the case in the European Alps – by enhanced nutrient input due to anthropogenic activities in the catchment [5].

4 Effect on the Thermal Regime

The management of reservoirs affects the thermal regime of downstream rivers in various – and often antagonistic – ways. To analyze water temperature changes, it is useful to distinguish between (1) the reaches immediately below the reservoir, (2) the abstraction section, and (3) the receiving river below the water release.

The effect of a reservoir is usually to warm the water. As the main inflows, consisting of cold water from snow- and glacier-melting, occur during the warmest months of the year, the heat uptake by the reservoir can reach enormous levels [10]. This is especially the case for turbid reservoirs (such as e.g. Mauvoisin or Grand Dixence) where particle-laden river water is heavier than reservoir water and the inflows plunge to the deepest reaches of the reservoir, entraining surface water downwards and thereby keeping the surface layer cold. Clear water reservoirs, in contrast, keep their temperature of maximum density near 4°C, but intense sunlight in alpine areas reaches deep into the clear water and warms the reservoir surface to almost lowland temperatures. Despite the high altitude, these mountain reservoirs have similar temperature characteristics as natural lakes. If the reservoir outlet is near the surface, the downstream water becomes much warmer than natural. If the water withdrawal is deep, the temperature change is small and the main effect is the release of the warmed water after summer (Table 1).

In the abstraction section, the heat fluxes and thermal environment hardly change, but as a consequence of the diversion, the river is shallower than natural. As the rate of temperature change is inversely proportional to water depth, the abstraction section adjusts more quickly to ambient air temperatures. As a result, temperatures in abstraction sections are higher than natural during summer and

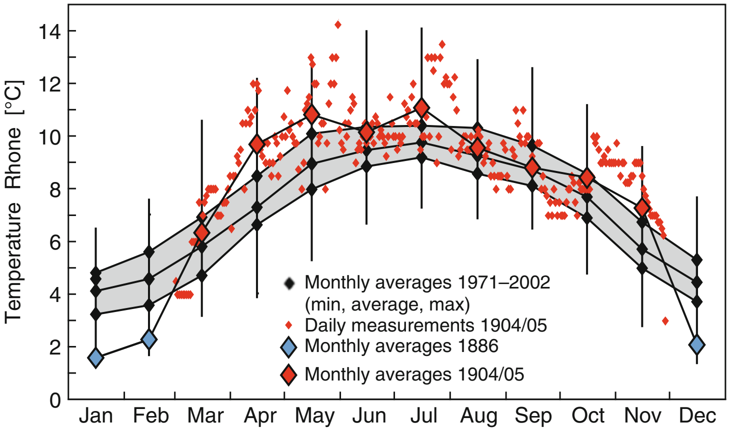

Fig. 5 The current River Rhone temperature regime (*grey, black*) as a result of damming (and partly climate change) compared with daily data (*red, blue*) from one century ago (1886 and 1904/5). The data has been collected at Porte du Scex by [19], by [18] and by [16] for 1886 (blue), 1904/5 (red), and in the last decades (grey, black), respectively. The *red dots* are daily values (afternoon). The *black bars* represent minima and maxima for the period 1982–2002. The hydropower operation has a "smoothing" effect on the temperature below the outlets in the River Rhone: In winter, the current temperature is about 2°C above natural (warming from pre-warmed reservoirs) and in summer about 1°C cooler (less riverbed friction; electricity production). The annual average remains almost constant. Details in [10]

lower than natural during winter, whereas the annual average is usually little altered [11, 12]. In both seasons, temperature amplitudes (e.g. daily) usually exceed – in some cases drastically – natural variations. If the water temperature is near 0°C, the abstraction-caused cooling can freeze the river bed.

Below water releases, in contrast, average temperatures are higher than natural in winter, and cooler than natural in summer (Fig. 5) – flattening the natural seasonal regime. Over the entire year, there remains usually a heat deficit, as the water passing through the turbines is cooler (electricity production) than in a natural river bed, where dissipation heats the water by 0.24°C per 100 m elevation decrease. The short-term temperature fluctuations – which can still reach a few °C – are also usually dampened due to increasing water flow at lower elevations. The frequencies of these fluctuations are directly coupled to the hydro-peaking operation of the power plant.

5 Ecological Implications for Downstream Lakes

As many alpine rivers – especially in the European Alps – contribute to lakes on their way to the lowlands and oceans, the effects of rivers are translated also to downstream lakes (Table 1). At high turbidity, the particle content determines the water density, whereas variations in temperature and salt content are often negligible. As particle concentrations increase with river flow (see above), the river water is denser at times of high discharge. During storms or flooding, the river water becomes heavier than lake water and forms deep intrusions, which may reach the deepest layers in the lake.

Loizeau and Dominik [1] compared observed deep intrusions in Lake Geneva with the discharge of the River Rhone and found deep downslope density currents whenever flow exceeds 400 $m^3\ s^{-1}$. As shown above, the occurrence of discharges greater than 400 $m^3\ s^{-1}$ in the River Rhone has sharply decreased in the 1960s as a result of damming. Whereas under natural river flows, this condition was met 55 times during the summer, the number of those instances has now declined to 15 per summer. As a consequence, heavily dammed alpine rivers show less deep plunging and therefore less deep undercurrents that bring substantial amounts of dissolved oxygen to the deep waters of peri-alpine lakes. In the example of Lake Geneva, the deep density currents not only re-oxygenate the oxygen-depleted deep water, they also support the deep convective mixing in the following winter by warming the deepest layers. Therefore, these deep density currents have an important ecological function for these peri-alpine lakes.

As shown in the Lake Brienz case study [13], many more subtle alterations occur in downstream lakes as a direct and indirect effect of the modified regime of water, nutrients and light. Whether such modifications have substantial effects on plankton [14] or other ecological indicators cannot always be clearly answered [13].

6 Mitigation Possibilities

As shown by the previous examples, alterations by damming to the aquatic environment are unavoidable. However, planning and thoughtful management provides many opportunities to reduce or even eliminate unwanted adverse effects.

(1) Mitigating the seasonal water flow is not realistic in large dams, as it would require a downstream reservoir of similar volume. However, short-term hydropeaking fluctuations can be levelled with a below-turbine retention reservoir by buffering daily water releases. Also, the frequency of extreme discharges (at the low and high flow ends) cannot be regenerated at reasonable costs. However, periodic flooding and dynamic residual flow management [15] become increasing viable (see also the example of the Colorado River).
(2) If reservoirs have a high ecological value, as in dammed former lakes (see example above), the lake-internal hydraulics can be altered by curtains or inflow/withdrawal structures. Thereby, organic matter, such as plankton, can be retained and nutrients can be guided through the water body in a more natural way.
(3) Withdrawal towers in dams allow corrections to the altered thermal regime without much additional effort. Especially in clear water reservoirs, withdrawal towers allow managers to choose water at its natural temperature and to reduce artificial seasonal temperature shifts. Again, retention basins have the same

effect on temperature as on flow, and therefore daily hydro-peaking fluctuations can be completely removed.

(4) Particle trapping is also a substantial economic issue, as in many countries the siltation time scale is relatively short (about 100 yr on a global scale). In alpine reservoirs this time scale is much longer, and measures to mitigate siltation had low priority. However, in the future, sediment management, including removal, will be unavoidable. An occasionally used method is to divert the early arrival of a turbid flash flood past the reservoir, so that most of the sediment is delivered to the downstream river. Also, sediment removal and downstream re-deposition is periodically applied, especially when the downstream river suffers from erosion.

(5) In remote reservoirs not affected by anthropogenic pollution, particle trapping can reduce the available nutrients. In the Canadian Rocky Mountains, fertilizer has been added to lakes and rivers in the form of liquid ammonium polyphosphate with the goal to stabilize nutrient concentration at historical levels [5].

These few examples demonstrate, that there are many possibilities to mitigate – at least partly – the adverse effects of damming. Needless to say, they all come with costs, which must be balanced with the precious natural value of alpine aquatic ecosystems.

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A Plea for the Restoration of Alpine Rivers: Basic Principles Derived from the "Rhone-Thur" Case Study

Armin Peter

Abstract Hydropower use and settlement development adversely affected Alpine rivers and streams. The majority of Swiss and Austrian electricity is produced by hydropower plants, which impact the natural flow regime of running waters. In addition, channelization impairs the river morphology.

The resulting changes (alterations of the natural flow regime, lack of hydrological connectivity, deficits in morphology) significantly affect the ecology of rivers and streams, and cause major deficits. Restoration of hydrological connectivity and ecomorphological integrity is an urgent need for Alpine streams. The "Rhone-Thur" project developed guidelines for planning river restoration and formulated ten basic elements for carrying out successful projects. Inclusion of a reference system, baseline monitoring and a clear definition of project objectives are important steps at the beginning. The project scale and ecological improvements are also specified. Each successful project has to deal with the socio-economic aspects and include the stakeholders in an early project phase. Prediction of restoration measures and the alternative possibilities are helpful processes. After implementation, the core of each project is the evaluation of success and outcome. Comparison of appropriate indicator values before and after the restoration helps to classify the project objectives into different success categories: from failure to great success. However, great success is difficult to achieve and it is important to understand the outcome. Generally, river morphology recovers more quickly after restoration than biological communities (fishes) do. The degree of connectivity to intact neighborhood communities and the size of the restoration area influence the success of the restoration project. Possible restoration sites should therefore be prioritized within the watershed, and sites close to intact habitat with existing source populations provide higher success expectations. For Alpine ecosystems the restoration of a

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dynamic hydrological regime is a major task and a challenge for the future. This is mainly true for stream reaches with residual flow and river segments with hydropeaking problems.

Keywords River restoration, Monitoring, Evaluation of success, Dynamic hydrological regime, Stakeholder involvement

Contents

1 Introduction

Over the last few centuries, people have substantially modified rivers and streams. Human use of European rivers increased and the ecological integrity (the ability of ecological entities to self-sustain, [1]) was adversely affected. Rivers were channelized for the purpose of land reclamation (urban and agricultural), flood protection, navigation and hydropower use. As a consequence, rivers became fragmented and the river corridor was lost, floodplain land disappeared, and riparian buffers (the green sponges) were dramatically reduced. In the Alps, tourist development and flow regulation for hydropower use has had a major impact on habitat conditions and on the discharge of rivers and streams. Dynesius and Nilsson [2] document that almost 80% of the total discharge of the main European rivers is affected to a greater or lesser degree by flow regulation measures.

In the Alpine region, running waters are essential resources for hydropower production. In Switzerland, most of the 1,600 hydropower plants are located in Alpine regions and, in total, there are 2,000 sites with water withdrawal. Around 58% of Swiss and 68% of Austrian electricity is produced by hydropower plants. The resulting changes in the hydrological regime have significant effects on the ecology of running waters and cause a lack of hydrological connectivity and pronounced deficits that primarily affect ecomorphology, bedload, water quality, biodiversity and recreational use [3].

Because the terms "restoration" and "rehabilitation" are not used in a consistent manner in the literature, the two are used synonymously in this article.

2 Need for Action and Area of Conflict

In Switzerland, the ecomorphological quality of rivers and streams is a function of altitude: it diminishes as the elevation decreases.

Below 600 m a.s.l., 50% of all streams have morphological deficits ([4], [5], this volume). This is particularly true for densely populated areas, where 85% of streams have morphological problems. At altitudes between 600 and 1,200 m a.s.l., 26% of streams show inadequate morphological features, and 11% of streams between 1,200 and 2,000 m a.s.l. have similar problems. Above 2,000 m a.s.l., only 2% of streams show inadequate morphological quality. In addition, longitudinal connectivity of Swiss rivers is markedly impeded by artificial barriers. Incised river channels and decoupled tributaries are a further crucial problem for connectivity. The average free-flowing stream reach between artificial barriers (> 50 cm) is only about 750 m. Although the morphological situation in Alpine rivers is less dramatic than on the Swiss Plateau, there is an urgent need to restore hydrological connectivity due to the number of water withdrawal sites and hydropeaking situations.

Restoring ecological integrity in rivers is a challenging task, and there is an urgent need for action. To gain the best ecological benefit for the funds invested, guidelines for planning successful projects are needed. The "Rhone-Thur" project aimed to define guidelines and tools for carrying out successful river restoration projects, and facilitating the transfer of knowledge between science and practice. Despite the importance of river ecosystems, river restoration projects are as much a social undertaking as an ecological one [1]. Restoration projects are carried out in a complex context; ecological, economic, social and political aspects must be taken into account and stakeholders involved at an early stage. In addition to restoration needs, many Swiss rivers also need improved flood protection. Because important synergies exist between flood protection and river restoration, this momentum and related benefits should be fully utilized: the utmost concern is to provide more space for the river.

3 The Basic Elements of River Restoration Projects

As an outcome of the "Rhone-Thur" project, ten basic elements of restoration projects can be formulated. To carry out successful river restoration, the complex character (ecology, flood protection, society) of these projects must be considered.

Restoration projects start with careful and comprehensive strategic planning (definition of reference systems and baseline monitoring), followed by a preliminary survey (defining of project objectives and possible restoration measures). Decisions concerning restoration measures require predictions of the possible consequences of management alternatives. We therefore developed an integrative river rehabilitation model [6] that predicts the hydraulic-morphological situation after measures have been completed, and subsequent changes in the aquatic and related terrestrial ecosystems have occurred.

The next phases are detailed planning of restoration measures, planning of project assessment and project implementation. The final phase, utilization, includes maintenance measures, evaluation of project success, adaptive management and the communication of results. Throughout all phases of the project, the general political and societal framework conditions are taken into account. In short, project managers must deal with complex processes and consider the contextual analysis [7].

Using the process sequence in Fig. 1, ten key elements of a restoration project can be formulated. These are considered below. The case study of the widening of the River Thur is discussed as an example. Widening of the River Thur at Schaeffaeuli (Switzerland) was carried out between 2001 and 2002. The river bed was widened from 50 to 100 m on both banks, along a river length of 1,500 m [8] (Fig. 2).

The planned river restoration (river widening) of the River Thur at Weinfelden-Buerglen is also integrated as a case study in the discussion of these ten elements.

3.1 Inclusion of a Reference System

To know the extent to which a river deviates from natural conditions, data from a reference system are indispensable. Ideally the reference is derived from a historical reference source or, if this is not possible, relatively undisturbed or already recovered reaches can be used to define reference conditions. Finding reference sites for large rivers is particularly problematic. It may make sense to use a severely impaired river as a reference for conditions to "move away from" [9]. However, using impaired sites as a reference often produces too optimistic assessments of the ecological success of restoration measures. It is easier to find a reference site for the Alpine area than for greatly altered lowland rivers. In general, historical information is ideal for establishing reference conditions. Historical maps are often easy to find and help to reconstruct the natural morphological situation, but ecological data are lacking in the majority of cases. Here a comparison with a theoretical reference system (analytical or process-based approach from concepts or classification systems) or an undisturbed, similar river (with the same general characteristics) may be an acceptable substitute for the lacking data. For the River Thur, we had historical fish community data from Wehrli [10].

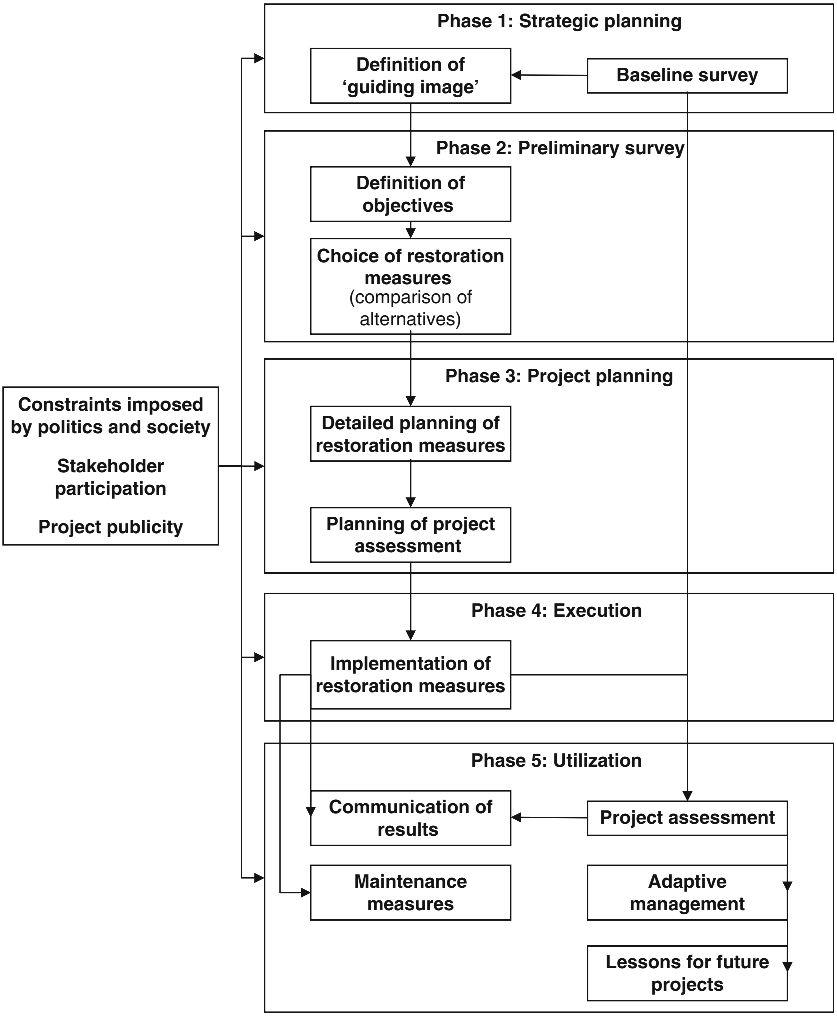

Fig. 1 Ideal procedure of restoration projects (from Woolsey et al. [3]), with permission from Wiley-Blackwell, Oxford UK

3.2 *Baseline Monitoring*

Project assessment starts at a very early phase by planning the baseline monitoring, which determines the pre-restoration ecological and morphological state. It characterizes the existing biota, chemical and physical conditions for planning or future comparisons [11], and identifies problems that require restoration measures. Unfortunately, many projects that have a monitoring approach fail to carry out baseline monitoring.

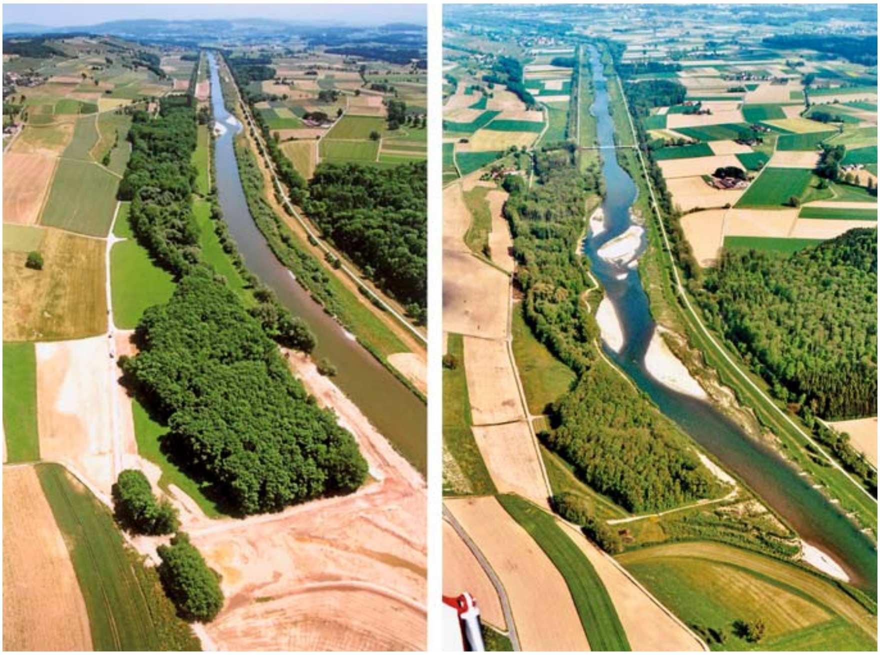

Fig. 2 The widened river bed of the River Thur at Schaeffaeuli (before and after the restoration). Photo: C. Herrmann, BHA team, Frauenfeld

Baseline monitoring can be combined with a "before-after" design, where data are collected both before and after treatment and thus replicated in time rather than space [11]. A "before-after control-impact" (BACI) design is often used. The control site is evaluated over a period and compared to the monitoring of the treatment site. A BACI design allows temporal variabilities in the control and treatment site to be compared.

If "before-treatment" (pre-restoration) monitoring cannot be performed, it is still possible to use a "post-treatment" design. This monitors the restored reach and untreated reaches that resemble the treated reach after the restoration was carried out. Roni et al. [11] discuss two types of post-treatment design: intensive post-treatment, in which multiple years of data are collected at one or more paired control and treatment sites; and extensive post-treatment, in which many paired treatment and control sites are sampled once each over a 1–3-year period. Whichever design is used, the monitoring will depend on available financial resources, but should reflect the restoration objectives.

3.3 Synergies with Flood Protection

Flood protection and ecologically oriented river restoration have one common goal: giving more room to the river. Basically, river channelization cannot be reconciled with the ecological goals of restoration projects. Flood protection that includes ecological components tries to give the river appropriate room. Local river widenings are good examples of giving the river more room, thus satisfying ecological as well as river engineering needs. In urbanized areas in particular, the availability of space is very limited and the river must be channelized and often between levees. In this case, it is very important to maintain minimal functioning; i.e., the riverbed should provide sufficient structure and longitudinal connectivity. Vegetation plantings within the riverbed could help to develop sparse riparian vegetation that provides cover for biota. The replacement of artificial weirs by ramps will support the migration of fishes along the river corridor.

Although the local widening project at the River Thur was primarily a flood protection project, ecological objectives were included in the goal-setting process during the entire planning process.

3.4 Definition of Clear Project Objectives

Further project evaluation (post-project appraisal) should be based on project objectives. Therefore, the main objectives must be explicitly defined in the preliminary study of any project. For many restoration projects, no clear objectives were ever formulated and consequently it is not really possible to assess their final success. The objectives should include elements of the environment/ecology, the economy and society. According to Reichert et al. [12], objectives can be divided into fundamental objectives (directly related to what a decision maker would like to achieve) and means objectives (leading to the accomplishment of fundamental objectives). Fundamental objectives are structured hierarchically. In the Rhone-Thur project, we used a hierarchy of fundamental objectives [12,13] that can be used for other restoration projects (Fig. 3).

Attributes are formulated for each objective in order to clarify the degree to which the detailed objectives have been fulfilled. As an example, the following attributes were defined for the objective natural river morphology and hydraulics, [12]: coefficient of variation of water depth, and coefficient of variation of flow velocity.

3.5 Project Scale: Priorities from a Catchment Perspective

River systems are fluvial networks of a population of channels and their confluences [14]. Restoration measures carried out on short river reaches generally have only a limited effect. Consideration of the catchment scale in river restoration programs marks a shift from the application of reach-based engineering principles towards an adoption of ecosystem-centered, adaptive and participatory approaches to river management [15]. Taking into account the catchment as a whole is indispensable

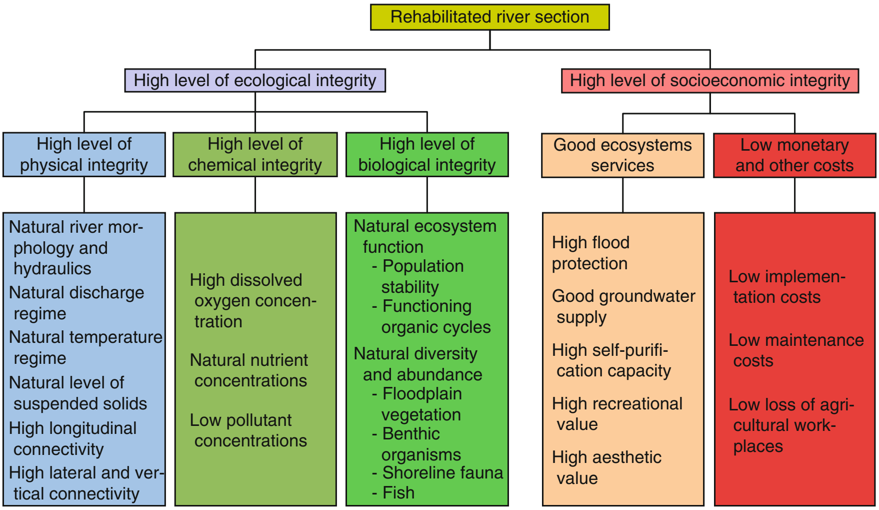

Fig. 3 Objectives hierarchy for river restoration projects developed in the Rhone-Thur project (from Reichert et al. [12]), with permission from Elsevier, Oxford

for the ecosystem-centered approach. For local measures, processes occurring in rivers and streams must be considered at the catchment scale [16]. One of the main problems faced by restoration projects is restoring longitudinal (upstream–downstream linkages), lateral (interactions between the channel and the adjacent riparian system) and vertical (interaction between channel and groundwater) connectivity.

Many projects have not successfully restored the biological community, due to inherent isolation problems and low or absent connectivity of the restored sites. The longitudinal, lateral and vertical dimensions of the whole river should therefore be considered early in the planning phase of the project. In fragmented rivers, recolonization processes are impeded or take place very slowly, and restored structures are not recolonized. Restored sites should have sufficient recolonization potential and connectivity. The reaction of the fish community in the River Thur after restoration is an interesting example. Although the habitat conditions were distinctly improved after restoration, the fish community has not responded: neither the total fish species number nor the total abundance of fish has increased. Six years after completing the restoration work, fish have still not responded to the observed change in habitat diversity [17]. Recolonization success depends on the degree of habitat isolation. In order to understand the lack of response of fishes to the restoration, ecological conditions for the whole River Thur must be taken into account; e.g., 65% of the lower 90 km of the Thur is morphologically classified as artificial or strongly impacted. Therefore, the restored site must be considered as an isolated reach that lacks connectivity to source habitats (biological hotspots of the river system). Depending on connectivity and nearness to intact neighboring communities, recovery time for fishes may take years or even decades [18]. Recovery processes depend on landscape, stream network conditions and related processes. Recovery is probably faster in Alpine rivers than for non-salmonid rivers in the lowlands.

3.6 Specifying the Ecological Improvements

Ecological restoration includes measurable changes in physicochemical and biological components [9]. The formulation of ecological objectives is an essential part of each project. This principle should also be applied to projects with a main focus on flood protection and improvement of hydrological connectivity in Alpine rivers. Examples of the formulation of ecological improvements include increasing structural diversity, improvements in hydrogeological dynamics (near-natural discharge regime), increasing ecological resilience of the river or stream, and restoration of the biological community.

To formulate ecological goals in ways that are more understandable to people living in the catchment, flagship species with special public appeal are often used. For the Thur, the nase (*Chondrostoma nasus*), which has a critical conservation status in Switzerland, was used to draw people's attention to ecological river restoration projects and served as a flagship species. Another flagship species was the little ringed plover (*Charadrius dubius*) and the German tamarisk (*Myricaria germanica).*

3.7 Including Socio-Economic Aspects

Integrating socio-economic aspects into project planning is a cornerstone for a successful project, but is also a challenging process for the project team. Stakeholders strongly influence the decision-making process. The project management team interacts with stakeholders, who can be divided into two groups: those from public administration and non-governmental stakeholders [7]. For the planning of the restoration project on the River Thur at Weinfelden-Buerglen, the following stakeholders were identified and are listed in order of increasing ability to influence the project and alternatives: recreational, forest rangers, federal administration, industry, environmental, agricultural representatives, communities, and regional (cantonal) or state (federal) administrations. Stakeholders may have different interests, and the main goal of the process of interaction is to find consensus. Multiple criteria decision analysis supports consensus finding and was used for the River Thur by Hostmann [19] by ranking different options for the different stakeholder groups.

3.8 Predicting the Effects of Restoration Measures

Although it is difficult to predict the outcome of restoration measures, this challenge must be tackled. Restoration measures considerably affect the hydromorphological and hydraulic features of a river. Therefore, the most relevant predictive task is to estimate hydraulic and morphological changes and their ecological impacts [12]. Schweizer et al. [6] developed an integrative model that was applied to a reach of the River Thur where a restoration project was planned. The morphological and hydraulic consequences of a proposed widening of the river were predicted. The model forecasted whether the resulting channel pattern would be straight, alternating or braided, depending on the width of the river after the restoration. Predictions of the subsequent depth and velocity of the river were also made [20]. In addition to hydraulic models, the following biological models were developed in our Rhone-Thur project: vegetation [21], benthic zone, the shoreline community, and fish [12]. Finally, an economic model predicted the impact of restoration activities on the local economy. Applied to the planned restoration at Weinfelden-Buerglen on the River Thur, each CHF 1 million expenditure per year yielded an extra eight full-time employment equivalents, and an increased output of CHF 1.4 million was estimated [22]. Careful predictions depend on the quality of the models used, and account for the credibility of planned restoration works.

3.9 Evaluation of Restoration Alternatives

Restoration measures and alternatives are formulated in the planning process. Several measures may be appropriate for increasing structural diversity (local river widening, creating side channels or backwaters, installations of instream structures, etc.). Restoration alternatives should be included in the analysis process, and ranked after predicting the outcomes of the various formulated alternatives. The ranking is an outcome assessment and contributes significantly towards resolving conflict when deciding between different options. In addition, it is an objective basis for discussion, an elaboration of consensus, and a support of the learning process [7].

3.10 Evaluation of Success and Outcome

In addition to the implementation process, evaluation of the project's success is the core of a restoration project. However, many projects do not take this final check, which is a prerequisite for adaptive management, into consideration. Bernhardt et al. [23] found that only 10% of project records (out of 37,099 projects) indicated that any form of assessment or monitoring was carried out. In most European countries, the estimated proportion of success evaluation is in the same range or even lower. Project evaluation through systematic data collection serves to verify whether the project objectives that were defined during the planning phase have been achieved [24]. Therefore, project evaluation is the process of examining the extent to which the defined objectives were achieved. Results of the baseline monitoring (initial conditions) are compared with the results after the restoration measure has been implemented. In the Rhone-Thur project, we developed a tool [3] for project evaluation, proposing 49 different indicators. Comparison of indicator values before and after the restoration results in a classification of the project objectives into five groups: (1) deterioration/failure; (2) no change; (3) small improvement/small success; (4) medium improvement/medium success; and (5) great improvement/great success. Appraisal of success therefore refers exclusively to the defined project objectives. However, an approximation to a reference system is achieved only if a good number of project objectives can be allocated to group 5 (significant improvement/major success). In order to compare the indicators, it is necessary to standardize them into non-dimensional values. Our indicator values lie between 0 and 1 and represent the degree of naturalness of, or the degree of satisfaction with, the relevant indicator.

The River Thur at Schaeffaeuli was widened from 50 to 100 m on both banks over a stretch of 1.5 km. The main goal of this restoration project was flood protection; the second goal was ecological improvement; and the third, the provision of recreational space. To measure the success of restoration in the Thur, 11 indicators were analyzed [17]. These indicators are listed in Table 1. Table 1 shows the standardized indicator values before and after restoration and classifies the success of the indicator into categories.

Habitat conditions showed distinct improvement (indicators 6, 9, 10), while for the recreational use there was an overall moderate success. However, the result was less promising for the direct ecological indicators (4, 5, 11). The two fish indicators used showed no improvement, while the vegetation indicator suggested minimal success. Even the latest monitoring results from summer 2007 and winter 2008 [25] showed no change in the situation. Therefore, 6 years after the completion of the river widening, fish have still not responded to the change in habitat diversity. In

Table 1 Standardized indicator values before and after restoration in the local river widening of the Thur River at Schaeffaeuli

Indicator	Value before	Value after	Success of indicator category
1. Number of visitors	0	1	
2. Variety of recreational opportunities	0.6	0.8	Medium success
3. Public site accessibility for recreation	0.4	0.7	
4. Fish species abundance and dominance	0.4	0.4	No change, failure
5. Diversity of ecological guilds of fish	0.4	0.4	
6. Variability of visually estimated wetted channel width	0	0.7	Medium success
7. Project cost: adherence to budget		1	Great success
8. Clogging of hyporheic sediments	1	1	No change
9.Width and degree of naturalness of riparian zone	0	1	Great success
10. Degree and type of anthropogenic modification	0.2	1	Great success
11. Succession and rejuvenation of plant species on floodplains	0.2	0.4	Small success

order to interpret this result, the ecological condition for the whole River Thur must be considered. Some 65% of the lower 90 km of the Thur is classified as morphologically artificial or strongly impacted. Recolonization processes can take place very slowly. For instance, Dunham et al. [26] reported that this process probably takes place over long time scales (>10 years).

The overall success of the local river widening project in the Thur is medium, but there are distinct differences between indicators. Generally, river morphology recovers more quickly from press disturbances (chronic disturbance affecting rivers in a subtle way, e.g., river channelization) than do fish. The degree of connectivity to intact neighboring communities and the size of the restored area may significantly influence the success of the restoration. In addition, potential restoration areas must be selected by prioritization analysis. River network maps documenting physical heterogeneities and biological hotspots in the river system may support the difficult search for the right restoration sites, which deliver the greatest benefit to the entire course of the river or catchment.

Because indicators for monitoring should be selected according to project objectives, each project has an individual set of indicators belonging to different indicator categories. In the evaluation procedure developed for the Rhone-Thurproject, 17 different indicator categories with a total of 49 indicators were proposed [3].

4 Conclusions

The complete restoration of rivers to their original condition is often impossible. However, rehabilitated riverine ecosystems can develop into natural, self-regulating systems. It is also desirable that restored reaches require no further maintenance after the measures have been completed [27]. The proposed ten elements of restoration projects may be applied to most future restoration plans and are also suitable for Alpine streams and rivers. However, for Alpine ecosystems, the restoration of a dynamic hydrological regime is the main task and will be a future challenge, essentially for reaches with residual flow and river segments with hydropeaking. Significant restoration efforts are needed for rivers with degraded morphological and hydrological conditions. Habitat and structural fixes alone will not restore the biological communities unless an appropriate flow regime is established because fundamental riverine functions depend on complex processes between hydrology, geomorphology and biota. Special indicators must be taken into account in the analysis and restoration of the flow regime in rivers with hydropeaking. Meile et al. [28] discuss possible mitigation measures for rivers affected by hydropeaking. The construction of retention basins is one of the most promising mitigation actions, but requires substantial financial resources and the necessary land available in the catchment at the right place.

Applying the ten proposed elements will ensure good project management and a high certainty of success for implemented restoration measures. Including an ecosystem-centered approach to prioritize possible restoration sites within the watershed [29] and setting a sequence of restoration actions will further increase the chances of success. Restoration projects should start near intact habitats with source populations and increase substantially the areas of good quality habitats with an intact connectivity.

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Water Management Challenges in Himalayan Watersheds

Hans Schreier

Abstract The Middle Mountains in the Himalayas contain some of the most intensively used watersheds in mountain regions of the world. As shown in this case study increased climatic variability is leading to widespread shortages of drinking and irrigation water and land stability, accelerated erosion, sediment transport and flooding are the main issues during storm events. Conservation farming to prevent soil erosion is a major challenge particularly on sloping agriculture. Degraded areas are the main source of sediments and rehabilitation efforts that focus on using nitrogen-fixing fodder trees, planted in hedgerows on sloping terrain, are proving to be an effective adaptation technique. Water harvesting and storage in ponds and cisterns during the monsoon season is another effective adaptation technique that enables the production of food via low-cost drip irrigation during the dry season. Reducing the land use intensification and population pressure on the marginal lands in the headwaters of these watersheds is likely to be the most effective coping methods in view of increased climatic variability.

Keywords Adaptation Techniques, Climatic Variability, Conservation Farming with Fodder Trees, Erosion, Low-Cost Drip Irrigation, Water Harvesting, Water Shortages

Contents

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1 Introduction to the Key Regional Water Issues

The water resources in the Himalayan mountain region are characterized by extremely high variability both regionally and seasonally. Some of the highest recorded rainfalls occur in the eastern part of the mountain range, with up to 11 m of annual rainfall in Cherrapunji, India, while the western part is very dry with less than 115 mm in places like Leh, India. There is also a very steep non-linear altitudinal gradient with the largest amount of precipitation occurring in the first two mountain ranges as one moves from the Terrai lowland to the Siwalik Mountains. Between 65–80% of the annual precipitation occurs during the four months of the monsoon season and a distinction must be made between glacial-fed and rain-fed streams. It is also a very tectonically active zone, which makes it susceptible to frequent earthquakes and landslides

The great uncertainty in this mountain region is how people cope with increased climatic variability. Because the mountains are the source of water for some 25% of the global population, the area is intensively used for food production, and in contrast to the European Alps the scientific knowledge base is quite poor.

Many of the glacial-fed streams that originate from the high mountains are subject to glacial lake outburst floods (GLOF) [1, 2] and as global warming proceeds this will have devastating consequences for the people living in the lowlands of the Ganges plain who are already subject to annual monsoon flooding [3]. Of equal or even greater concern are the tributary watersheds in the Middle Mountains which are dominated by surface runoff processes from monsoon storm. These are the watersheds where the regional population density is the highest and where the land use intensification is among the greatest of any mountain region in the world. The focus of this case study is on the Middle Mountain watersheds in Nepal and India because: population growth is very high (>2.3%), most of the population is engaged in intensive subsistence agriculture on very steep slopes, all natural resources are under pressure, and increased climatic variability will potentially have devastating effects on the livelihood of the people.

Agriculture, which is not only the largest water user, requires more water and this land use is also emerging as one of the main contributors to water pollution. Many subsistence farmers have benefited from the green revolution, which resulted in the introduction of multiple annual cropping cycles (up to four-crop rotations per year). Access to high yielding crop varieties and fertilizers was facilitated through foreign aid, and expanding agriculture onto steep slopes has resulted in improved food production. Forests are an integral part of agriculture, supplying litter, fodder, and firewood, and are the major source of energy for the local communities. Agricultural intensification has helped to improve food security and has been made possible by intensive irrigation on many man-made terraces built on very steep slopes. However, the question needs to be asked if these high-input systems are sustainable given the extensive labor needed to maintain these terraces, the high demand for fertilizers, the problems associated with water pollution, and the availability of sufficient water resources. The risk of slope failure during storm events, and the emerging water constraints during the dry period are issues that will become more apparent as climatic variability increases.

Groundwater is very difficult to access because of lack of infrastructure and the presence of many uplifted and interbedded sedimentary and metamorphic formations. Most people rely on local springs, shallow groundwater, and surface water as their main source of drinking water. This means daily treks to collect and carry drinking water for people and animals over large distances. There is considerable evidence to suggest that acute shortages of irrigation and drinking water are already the norm in the dry season in many parts of the region [4–6]. Kathmandu, which is also located in the Middle Mountains, is experiencing extensive water shortages during the dry season as a result of very rapid population growth, limited capacity to store water during the monsoon season, and a highly neglected and vulnerable infrastructure.

Historic records show how devastating intensive monsoon storms can be in the Middle Mountains. In 1993 a storm that produced 540 mm of rainfall over a 24 h period and with rainfall intensities exceeding 70 mm h^{-1}, hit the Kulekhani watershed [7–9]. This resulted in hundreds of landslides and a sediment load of up to 500 T ha^{-1} per storm in the watershed (Fig. 1). This not only destroyed large tracks

Fig. 1 Impact of the 1993 monsoon storm in the Kulekhani Watershed which delivered 540 mm of rainfall within 24 h

of forests and agricultural fields but also reduced the storage capacity of the hydropower reservoir located in the lower part of the basin by one-third [9, 10]. Similar, highly localized extreme events have been reported at a frequency of 6–9 years in many other watersheds. The main concern for water managers in this region is the possibility that these extreme events are likely becoming more frequent or are even increasing in strength as a result of climate change. The question that needs to be addressed is how much adaptation is possible and what precautionary practices can be put in place to minimize such impacts.

2 Case Studies from Middle Mountain Watersheds

The Jhikhu Khola and Jarsha Khola watersheds are two typical Middle Mountain watersheds (Fig. 2) With no glaciers, where snowfall is very rare and only occurs at the upper elevation of the watersheds, and where most precipitation occurs during the monsoon season in the form of rain. This is followed by an extended dry season which often lasts for 5–7 months. The characteristics of the two watersheds are provided in Table 1.

The Jhikhu Khola watershed represents one of the most intensively used watersheds in the region and the Yarsha Khola watershed, somewhat less intensively used, should be considered more typical of the Middle Mountain region. The problems in the Jhikhu Khola are thus a reflection of what is in store if population growth and land use intensification continue to increase in these mountain areas.

Fig. 2 Location of the Jhikhu and Yarsah Khola Watersheds in the Middle Mountains of Nepal

Table 1 Characteristics of two Middle Mountain watersheds

	Jhiku Khola watershed	Yarsha Khola watershed
Size (ha)	11,000	5,300
Irrigated agriculture (%)	16	14
Rain-fed agriculture (%)	39	37
Forests (%)	30	32
Shrub and grazing land (%) (including degraded lands)	15	17
Elevation range (m)	700–2,300	990–3,030
Average annual rainfall (mm)	1,167–1,420	2,010–2,469
% of Average annual rainfall during monsoon season (4 months)	70–80	65–78
Average annual discharge (m^3 s^{-1})	1.45	2.2
Extreme min. and max. discharge (m^3 s^{-1})	0.01–33	0.08–14.3
% of rainfall resulting in discharge	32	62
Public water sources	319	215
Population density (people per km^2)	437	386

Sources: [6, 11–13]

3 Water Availability, Use, and Constraints

The water supply in both watersheds is primarily from rainfall, surface runoff, and springs that are entirely recharged by rainfall. This means that between late May and early September the watersheds receive too much water resulting in terrain instabilities, erosion, and flooding. After September there is continuous drying until late May with only the occasional occurrences of rainstorms. The historic rainfall records are of limited duration and apart from the extreme seasonal variability, it was not possible to show any evidence of changes in the rainfall pattern over the past 10 years [14].

Over an eight year period from 1993–2000 the discharge in the Jhikhu Khola varied by more than three orders of magnitude from an extreme low in the dry season of 0.01 m^3 s^{-1} to an extreme high of 33 m^3 s^{-1} during the monsoon season [14]. The variability in the Yarsha Khola was similar and ranged from 0.08 m^3 s^{-1} to 14.3 m^3 s^{-1} (over a 3 year period). From field observations there is evidence that the low flow stream discharge has declined over the past 15 years but the data logging equipment was not sufficiently sensitive to address the low flow conditions. At this point there is insufficient evidence to suggest that the decline in low flow is due to increased climatic variability, but there certainly has been a significant increase in water use for irrigation by the expansion and intensification of irrigated agriculture [12].

Evidence of the declining availability of freshwater over the past 15 years was obtained from a social survey conducted in 2000 in which 365 male and female residents were asked to list their greatest concern about access and sustainability of the resources they rely on [5]. Drinking water and irrigation water shortages were listed by more than 60% of the respondents as being critical issues, with an average of 40% suggesting that the availability during the dry season has declined.

4 Issues with Domestic Water Use

The average domestic water use in the two watersheds was estimated to be 21–23 L per day per person for drinking water sanitation and food preparation [6]. This is at the very low end of the minimum requirement of 20–40 L per day per person recommended for drinking water and sanitation alone by WHO [15]. None of the residents have piped drinking water in their homes. This means that for a family of four over 100 L of water are collected daily from the nearest spring. In the majority of cases the domestic animals are stall-fed and require on average 70 L per day. If they cannot be moved daily to local streams then additional water needs to be collected for livestock watering. Women and children are dominantly designated to collect drinking water and, as water shortages increase during the dry season, the workload to carry the water over long distances increases. It was estimated that the average time to collect the daily water requirement for a family is 16 minutes but some families have to spend up to 120 minutes per day to collect the water. While there appears to be an adequate supply of domestic water in terms of annual totals, it is in the dry season where the problem is critical. There is clear evidence of some wells drying up during that period.

The second issue is the water quality. On average some 18 households depend on a single domestic water source, which is dominantly an open spring or local taps fed by small unlined reservoirs. Only 2–3% of the people have resorted to constructing shallow dug wells near their houses but these frequently dry up during the dry season. Since the infrastructure around the springs is minimal, sediment and pathogen problems are widespread. Almost all samples analyzed from 31 sources had positive counts for fecal coliform, with higher values during the monsoon season [6]. This suggests that contamination from surface runoff is the main source. In addition, eutrophication is evident in shallow wells and ponds because of the very intensive use of fertilizer and manure in agriculture.

5 Issues with Irrigation Water Use

In order to meet the food demand of the rapidly expanding population in the Jhikhu Khola watershed agricultural intensification has moved from an average of 2.0 annual crop rotations to 2.7 rotations between 1990–2000, and some fields are under four annual rotations [12]. This was only made possible due to access to short-growing season crop varieties, intercropping, intensive fertilization, and irrigation. The emphasis on the green revolution has resulted in the production of excess food that is exported out of the watershed during those periods when there is sufficient water. However, with this intensification it is very challenging to provide water to the thousands of terraces that are irrigated in each watershed. The traditional community-based irrigation systems are impressive but there are very large losses because few of the delivery channels are lined and the loss due to evaporation is high because all terraces are flood irrigated. The community-based indigenous irrigation organizations are well organized and rapidly respond to individual terrace failure. If an individual terrace fails during the monsoon, this can result in a very rapid cumulative effect that can create massive landslides and destroy a large part of the entire system of hundreds of terraces. Maintaining nutrient balances and soil quality in these intensive systems is very difficult [16] and dry season water shortages are now widespread resulting in emerging water conflicts.

6 Issues with Erosion and Sediment Transport

This seasonally extreme climatic regime poses a significant challenge to agricultural and forestry management in these mountain watersheds. Cultivating these extremely steep slopes is particularly difficult because of the labor involved in maintaining terraces and minimizing soil erosion. As shown in Table 1, between 14 and 16% of the watersheds are irrigated on series of individually terraces. 37–39% of the slopes are under rain-fed agriculture with minimal terracing, and 30–32% of the watersheds are in community forests which are heavily used for litter, firewood, and fodder collection. Up to 7% of the watersheds are in a highly degraded state with little vegetation and soil cover and widespread rills and gullies. It was estimated that these degraded areas in the Jhikhu Khola contribute up to 40% to the annual sediment load [17]. Some of the eroded soils are captured on terraces through deposition via the irrigation water. While this adds new sediments to the terrace and helps soil development, the nutrient contribution from these degraded sites is small. In contrast if the sediments originate from fertilized rain-fed non-terraced fields then the contribution to production by added nutrients in the sediments can be significant. Brown et al. [16, 18] and Von Westarp et al. [19] determined that between 6–11 kg ha^{-1} of Nitrogen and 2–3 kg ha^{-1} of P_2O_5 is added annually on to irrigated fields from sediment deposited via the irrigation water from rain-fed agriculture. However, the overall losses through the system are of course much more substantial.

There is little erosion from the terraced system but rain-fed and forested areas are prone to erosion during critical times of the year. Farmers do a remarkable job in minimizing erosion by many different conservation measures, but during unusually extreme storm events a large amount of erosion and sediment transport takes place. Research from erosion plots on rain-fed sloping agricultural land between 1990 and 2000 showed that annual erosion rates averaged 14.5 t ha^{-1} (range 2–37 t ha^{-1} per year) [14, 20, 21]. The average annual erosion rate from degraded sites was 20.5 t ha^{-1} per year (range 6–39 t ha^{-1} per year). The degraded plots produced approximately one-third more sediments than the agricultural plots [20]. The contribution from forests is considerably less but no quantitative information was available for comparative purposes.

The farmers are well aware of the erosion risk and they try to stabilize soils after failure by dewatering the sites and by conservation practices that include the use of bunds, hedgerows, and detention channels. However, cultivating these slopes in a monsoon climate is always a challenge.

Contrary to common belief the greatest erosion risk for rain-fed agriculture is not during the peak of the monsoon season, but in the pre-monsoon season when the farmers prepare the fields for cultivation. This happens after an extended dry period and after the first and second monsoon storms arrive to facilitate cultivation. At that time of the year there is no vegetation cover to protect the soil and any intense storm during that period has devastating effects. Individual storms during this period can remove up to 10.7 t ha^{-1} of soil per storm in the upland catchments [17]. The timing of cultivation thus poses the greatest risk because the variability and intensity of these early storms cannot easily be predicted. Planting a cover crop than can survive during the dry season and practicing minimum tillage are some of the options possible to minimize these erosion risks.

In summary, the water challenges in these rain-dominated Middle Mountain Watersheds are large and while the inhabitants are well aware of the risk, and while they have developed effective adaptation strategies, it is difficult to reduce these risks particularly if population growth and land use intensification proceed. All of these problems have been studied with an emphasis on the natural variability that has occurred over the past 20–30 years and the rapid changes in land use that have occurred over the same time period. What is now of additional concern is how increased climatic variability will affect these watersheds. How will the risk change with increased climatic variability? We are unable to separate the combined effects of land use and climate change because they are occurring at the same time. However, we can speculate on possible impacts and develop adaptation scenarios, which are the topics addressed in the next section.

7 Impact of Increased Climatic Variability

Paleo-records [22] suggest that the monsoon storms have very rarely failed but the start of the monsoon period has always been highly uncertain. A late arrival of the monsoon season can be devastating because this extends the dry season when water shortages are widespread. More important is the frequency and intensity of the monsoon and pre-monsoon storms. If the type of storms experienced in the Khulekhani watershed in 1993 is any indication of what might be in store for the future, then it will be a major challenge to sustain the type of intensive agricultural systems practiced in these watersheds. While many of the terrace systems survived better than the sloping agricultural land, it took 3–5 years before slope failures could be sufficiently stabilized to recreate the terrace system and this required enormous human effort [9]. However, the mobilized sediment is proving to be a long term management problem because it not only clogs up many of the irrigation systems and changes stream channels, it also impacts the sediment transport processes downstream. The sediment load remains high for many years after the storm and it might take a very long time before the system reaches levels that approach pre-storm level This has large consequences for the viability of future hydropower reservoirs and provides enormous challenges for the people living downstream of these watersheds.

8 Adaptation Strategies for Climate Change

Himalayan farmers are used to living in a highly risk prone environment and have been very progressive in adopting new and different technologies. In order for new innovations to be effective the focus has to be on efforts that reduce the already excessive workload necessary to maintain these very intensive agricultural systems.

8.1 Adaptations to Cope with Extended Dry Periods

Effective adaptation strategies in anticipation of extended dry periods include water harvesting and storage during the monsoon season, when an excessive amount of water is available. This stored water can then be used during the extended dry season when water shortages are widespread. Attempts to collect roofwater as a source of drinking water have been explored in two of these watersheds [23]. While this is a viable option, not all roof surfaces are suitable for this practice and maintaining safe drinking water while it is being stored is another challenge. A more effective adaptation strategy is to store surface water runoff during the monsoon and then use it for low-cost drip irrigation during the dry season. Building low-cost underground cisterns was first thought to be the most water efficient option because it minimized evaporative losses. However, dug-out ponds to store monsoon water proved to be a much more desirable option, because in these tectonically active watersheds, cracks and subsequent leakages are common and the labor required to build and maintain these underground systems is much higher than maintaining water in ponds. These ponds are small and do not require much space but are capable of holding sufficient irrigation water to be used with low-cost drip irrigation systems for growing a vegetable crop in the dry season when food shortages are common and prices are high. Every day a 30–40-L storage tank is filled with pond water, which then moves sufficient water by gravity through the drip lines to the plants. This is a viable adaptation strategy that is cost effective and requires little labor [24]. This practice was taken up by several hundred women farmers in the Jhikhu Khola watershed between 2000 and 2004. In contrast to centralized large reservoir storage systems, the risk of pond failure is small and sediments can be managed effectively by building sediment traps before the runoff water enters the pond.

Another effective adaptation strategy is to improve the conventional gravity-fed flood irrigation practice which includes lining delivery channels, and reducing the amount of water applied and selecting less water consumptive crop varieties.

8.2 Adaptation to Increasing Storm Events

This is a much greater challenge and requires measures that often are beyond the capability of the rural population. One of the key problems is the need to rehabilitate badly degraded areas. Although only up to 7% of the watershed is heavily degraded and no longer in use, the erosion problem from these areas has very destructive effects downstream, where the land is heavily used for food production. These degraded sites have been abandoned a long time ago, they have little vegetation cover, are heavily gullied and the remaining soil and the exposed surficial material have very poor nutrient contents. Most of the degraded sites are on red soils that are geologically old, heavily leached, have high Fe and Al content, are devoid of phosphorus, and are very acidic (pH 3.8–5.2) [25]. Stabilizing these sites and establishing a protective vegetation cover is key to reducing the sediment problem downstream. This can best be established by planting hedgerows of fodder trees that can fix nitrogen and phosphorus (are mycorrhizal tolerant). More than 25 different fodder trees are native to the region and can readily be used for this purpose [26].

Developing hedgerows of fodder trees should also be practiced on rainfed agricultural fields on sloping terrain, since this is another effective adaptation technique that reduces the risk of slope failure and soil erosion.

However, if storm events become more severe and more frequent then there is clearly a need to reduce the land use intensification and the cultivation of some of the steeply sloping fields should be abandoned and converted back into forest cover. This would have additional benefits by improving the fuel, fodder and litter supply, a supply that is currently insufficient from the available forests that are heavily degraded in many parts of the watershed. At the same time the forest, if sufficiently rehabilitated, will retain moisture, reduce runoff, and protect soil erosion. This is obviously a very big challenge given the rapid population pressure in the Jhikhu Khola watershed. During the civil conflict over the past 5 years there has been a bit of an exodus of people, many of whom have become migrant laborers abroad. The financial return from abroad can have some positive impacts in that it will provide resources for improving the infrastructure in the watershed. These measures can move a long way to optimizing water use and reduce the pressure on the water resources. If used more widely they might also be effective in reducing the risk of more water shortages as a result of increased climatic variability.

Finally, an early warning system should be in place at the time of the most intensive storm events and community development in the hazardous lowland riparian zone should be discouraged. The communities in the mountain watersheds are well aware of these risks and few of the communities have developed in the most hazardous zone. However, it is the lowland population in the Terrai region that is at the greatest risk because they are impacted by events taking place in the Middle Mountains some 50–100 km away and are the recipients of cumulative effects. Predicting the timing of extreme flooding events and providing sufficient advance notice is a topic that needs considerably more attention and research.

9 Conclusion

The key focus in the Himalayan Mountains should be on improved water management and adaptation initiatives to address increased climatic variability. The region supplies water to more than 25% of humanity and is prone to extreme seasonal variability. The Middle Mountain zone is particularly at risk of increased climatic variability because in this zone population pressure and land use intensification is among the highest of any mountain region in the world and the scientific knowledge base is relatively poor. In this case study it was shown that the extreme seasonal variability is already causing major problems with land instabilities, sediment transport, and flooding during the monsoon period, and extended water shortages during the dry season. These problems are likely accentuated as a result of increased climatic variability. A number of adaptation strategies are proposed to reduce the water shortage problem during the dry season through different water harvesting techniques. The problems associated with storm events can in part be addressed by increased attention to the use of improved soil conservation methods using hedgerows of fodder trees interspersed into sloping agriculture, and major rehabilitation efforts on degraded land. However, extreme storm events pose a major challenge and few adaptation techniques will suffice in reducing the risk and downstream impact. For these cases what is needed is an advanced warning system and the development of a strategy to discourage community development in hazardous lowland zones.

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Index

G

H

I

J

K

L

M

Z