



FACULTY OF TECHNOLOGY

# **Hydropeaking Mitigation with Re-regulation Reservoirs**

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# **Abstract**

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In a scenario aiming to achieve Net Zero Emissions by 2050, hydropower is expected to play an increasingly significant role as a renewable and balancing power source. However, the prevalence of hydropeaking, a common occurrence in hydropower plants, is expected to intensify. Consequently, this will cause flow alteration and increase river ecological stresses. Hydropeaking has diverse ecological impacts on ecosystems, ranging from changes in habitat availability to alterations in species composition, population, and biodiversity, while also leading to water quality degradation. These impacts emphasize the importance of employing hydropeaking mitigation measures that consider the ecological impacts and environmental non-stationarity without increasing the operational cost of hydropower plants. In this study, we present an innovative engineering approach to mitigate the adverse impacts of hydropeaking on river flow regimes using auxiliary reservoirs termed as re-regulation reservoirs. The primary objective of re-regulation reservoirs is to reduce flow fluctuations resulting from hydropeaking by diverting and storing excess water during high flow periods and subsequently releasing it back to the river corridor during low-flow periods. Re-regulation reservoirs offer several advantages, such as greater flexibility and adaptability to varying environmental conditions, power, and water demand, without increasing the operational costs of power systems. The operation and definition of re-regulation reservoirs entail constraining various flow components, thereby limiting their threshold values. This study evaluated the operation and efficacy of a potential re-regulation reservoir using data from river systems. The investigation involved testing a variety of threshold values to assess the effectiveness of re-regulation reservoirs in mitigating hydropeaking impacts. The study developed a methodology and an open-access algorithm for operating re-regulation reservoirs,

establishing a hierarchy of conditions to restrict peak flow, minimum flow, up-ramping rates, and down-ramping rates. The findings demonstrated that re-regulation of reservoirs holds clear theoretical potential for regulating hydropeaking. Our results demonstrated that the correlation between flow component thresholds and the required re-regulation of reservoir volume varies depending on the river's flow pattern. Additionally, our findings underscore potential pathways to optimize the design of re-regulation reservoirs.

*Keywords: Altered Flow, Ecological Impacts, Hydropeaking Impacts, Mitigation*

# Tiivistelmä

Lyhytaikaissäännöstelyn lieventäminen uudelleensäätöaltilta

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Skenaariossa, jossa tavoitellaan nettonollapäästöjä vuoteen 2050 mennessä, vesivoimalla odotetaan olevan yhä tärkeämpi rooli uusiutuvana ja tasapainottavana energialähteenä. Vesivoimalaitoksissa yleisen lyhytaikaissäännöstelyn odotetaan kuitenkin voimistuvan aiheuttaen virtausmuutoksia ja lisäävän jokien ekologista raskautusta. Lyhytaikaissäännöstelyllä on monia erilaisia ekologisia vaikutuksia ekosysteemeihin ja paikalliseen hydrologiaan. Nämä vaikutukset korostavat lyhytaikaissäännöstelyn lieventämistoimenpiteiden tärkeyttä nostamatta kuitenkaan vesivoimalaitosten käyttökustannuksia. Tässä tutkimuksessa esittelemme innovatiivisen teknisen lähestymistavan lyhytaikaissäännöstelyn haitallisten vaikutusten lieventämiseksi jokiuomissa käyttämällä apuvarastoaltaita, joita kutsutaan uudelleensäätöaltiliksi. Uudelleensäätöaltilaan ensisijainen tavoite on vähentää lyhytaikaissäännöstelyn aiheuttamia virtauksen vaihteluita ohjaamalla ja varastoimalla ylimääräistä vettä runsaan virtauksen aikana ja päästämällä sitä myöhemmin takaisin jokiuomaan pienempien virtausjaksojen aikana. Uudelleensäätöaltilat tarjoavat useita etuja, kuten suuremman joustavuuden ja mukautuvuuden vaihteleviin ympäristöolosuhteisiin, sähkön- ja vedentarpeeseen ilman, että käyttökustannukset nousevat. Uudelleensäätöaltilaiden toiminta ja määrittely edellyttävät eri jokivirtauskomponenttien rajoittamista, mikä rajoittaa niiden kynnsarvoja. Tässä tutkimuksessa arvioitiin mahdollisen uudelleensäätelyaltilaan toimintaa ja tehokkuutta käyttämällä todellisia tietoja lyhytaikaissäännöstelyn vaikuttavista jokiuomassa. Tutkimuksessa testattiin erilaisia kynnsarvoja, joilla arvioitiin uudelleensäätelyaltilaan tehokkuutta lyhytaikaissäännöstelyn vaikutusten lieventämisessä. Tutkimuksessa kehitettiin metodologia ja avoimen pääsyn algoritmi uudelleensäätöaltilaiden käyttöä varten,

määrittäen huippuvirtauksen, minimivirtauksen, ylös- ja alavirtausnopeuksien rajoituksia. Tutkimuksen löydökset osoittivat, että uudelleensäätelyaltilla on selkeä teoreettinen potentiaali lyhytaikaissäätelyyn vaikutusten vähentämiseen. Tuloksemme osoittivat, että virtauksen kynnyksiarvojen ja vaaditun uudelleensäätöaltaan tilavuuden välinen korrelaatio vaihtelee joen virtauksien mukaan. Lisäksi havaintomme korostavat mahdollisia tapoja optimoida uudelleensäätelyaltaiden suunnittelua.

## FOREWORD

I am pleased to present this master's thesis titled "Hydropeaking Mitigation with Regulation Reservoirs". This research project has been an exciting and rewarding journey, allowing me to explore and contribute to the field of environmental engineering with a focus on water related issues.

I would like to express my heartfelt gratitude to my supervisors, Dr. Ali Torabi Haghighi and Dr. Hannu Marttila, for their invaluable guidance, unwavering support, and insightful feedback throughout the entire process. Their expertise and encouragement have been instrumental in shaping this thesis into its final form.

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Lastly, I want to acknowledge the academic community and the countless researchers whose prior work has laid the foundation for this study. Their scholarly contributions have enriched my understanding and provided a framework for my research.

This thesis represents the culmination of extensive research, analysis, and innovation. I sincerely hope that the findings and recommendations presented herein contribute to the ongoing efforts to address the impacts of hydropower on water resources.

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ABSTRACT

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## 1. INTRODUCTION

As of 2018, hydropower constitutes a significant proportion of the renewable electricity sources in the European Union (EU), contributing to 36% of renewable electricity and 10% of gross electricity production (Alsaleh et al., 2023). Due to its flexible nature as a power source, hydropower is expected to have an amplified role in balancing and stabilizing the power market (Ashraf et al., 2018). With the European Commission's proposal of the European Green Deal, aimed at achieving climate neutrality by 2050, these figures are anticipated to increase. Despite its flexibility as a power source, hydropower can cause abrupt changes in sub-daily river flows, which is referred to as hydropeaking. Hydropeaking is an artificial flow pattern in rivers that arises from the periodic discharge of water, resulting from the rapid shifts between increasing and decreasing power generation in response to the fluctuations in the power market by hydropower plants (HPPs) (Bieri et al., 2014). Therefore, modifying the power demand may indirectly impact flow conditions and potentially exacerbate river hydropeaking (Ashraf et al., 2022). Consequently, anthropogenically-induced flow variations are recognized as a significant threat to aquatic ecology (Bunn et al., 2002). Changing the natural flow conditions can impose ecological strain, as it is widely acknowledged that they play a critical role in maintaining the ecological integrity of rivers and their corresponding floodplains (Poff et al., 1997). Such alterations may also have detrimental effects on the processes that shape the physical habitats of streams, which are essential for the biotic and abiotic composition and properties of rivers (Schmutz et al., 2018; Bunn et al., 2002; Jelovica et al., 2022).

Hydropeaking is a phenomenon that results in significant variability in flow patterns, which can have a substantial impact on river ecosystems (Meile et al., 2011). This variability is more pronounced than that resulting from other flow variations, such as seasonal changes or daily fluctuations (Bejarano et al., 2018). Moreover, hydropeaking releases have substantial impacts that can be considerably larger than natural flows, resulting in significant alterations in water depth and velocity (Shen et al., 2010). These alterations can lead to changes in water temperature, depths and velocity, and overall water quality, thereby impacting aquatic organisms' survival and reproductive success, including fish (Auer et al., 2022; Austin et al., 2015). Additionally, the effects of hydropeaking on river systems extend to recreational usage. Regulatory bodies typically impose environmental restrictions on the operation of hydropower plants (HPPs), such as

establishing minimum environmental flows and flow change rate limits to reduce the adverse effects of hydropeaking. Nevertheless, these operational constraints may not effectively achieve their intended environmental objectives and could impose financial burdens on HPPs (Guisández et al., 2016).

HPPs have the potential to cause significant fluctuations in downstream flow regimes, with up to five hydropeaking events occurring daily in some cases (Schmutz et al., 2018), depending on the extent to which they contribute to meeting power market demands. Despite the known negative impacts of hydropeaking on ecosystems and water resources, there is a shortage of engineering solutions to mitigate the recurring occurrence of hydropeaking events effectively. This could be attributed to the complex interplay of physical, biological, and societal factors influencing river dynamics. Hence, a comprehensive understanding of sub-daily flow dynamics can aid in developing effective hydropeaking mitigation strategies, as recommended by the European Union (EU) Water Framework Directive (WFD) and by the Canadian Fisheries Act and the United States of America (USA) Clean Water Act. The objectives of this study are to; 1) Review the impacts of flow alterations; 2) Review the impacts of hydropeaking; 3) Review potential mitigation measures; 4) Examine the potential of deploying re-regulation reservoirs (RRR) downstream of HPPs to mitigate hydropeaking's environmental impact; 5) Design a model to determine the required volume of RRRs to shave the peak flow, increase minimum flow, and limit ramping rates; 6) Determine the required RRR volume for 24 stations downstream of HPPs in Finnish rivers.

## **2. RIVER FLOW**

River flow is the driver for processes that control and shape waterways' morphology and ecosystem dynamics. The increasing demand for water resources is causing a continuous river flow alteration resulting in major changes to hydro-morphological processes (Grant et al., 2003). In response, legislative requirements have shifted water resources management over time to focus on ecological aspects (Warren et al., 2015). This has prompted the “Environmental Flow” term to come into use when describing the quality and quantity of water required to sustain a healthy ecological status (Acreman et al., 2008, Tharme, 2003). Environmental flow is widely used in river management, despite it being challenging to quantify for complex ecological systems. Environmental flows are usually defined as a departure from a baseline state, such as Normal Flow (Warren et al., 2015).

However, the normal flow is subject to natural seasonal and climatic changes. Thus, it is crucial to maintain a strong scientific foundation of natural flows to ensure the ongoing success of environmental flows in the future. However, new challenges are emerging that necessitate revisiting the fundamental principles of the natural flow regime that underpin current e-flow practices. Additionally, it is essential to devote renewed efforts to developing a more robust ecological foundation to support a more accurate and predictive environmental flow science (Poff, 2018).

### **2.1. The natural flow regime (NFR)**

The natural flow regime (NFR) is the pattern and variability of water flow that occurs in the absence of human alterations. For several decades, it has been established in scientific literature that the NFR represents the optimal flow regime, which is essential for maintaining the functional integrity of river ecosystems. The NFR of a river varies on time scales ranging from hours to years and longer (Sofi et al., 2020). The natural variation of NFRs leads to substantial impacts on river organisms, as well as on the ecological settings and processes that shape the river system (Lytle et al., 2017; Lake, 2000). Additionally, river flow regimes display regional patterns determined by the river size, geology, topography, vegetative cover, and regional climate. Poff and Ward 1989, identified five key components of flow regimes that govern ecological processes: magnitude, frequency, duration, timing, and rate of change of hydrologic conditions. These components have since been widely acknowledged and incorporated into the study of river ecosystems, as they are essential for sustaining the biodiversity and ecological integrity of rivers (Rosenberg et al., 2000). They can be used to depict the entire spectrum of flows and distinct hydrologic events that are crucial to the integrity of river ecosystems, such as floods or low flows. By outlining flow regimes based on these components, the ecological implications of certain human activities that alter one or more components of the flow regime can be explicitly evaluated (Poff et al., 1997). Nonetheless, these components engage in intricate interactions to govern geomorphological and ecological phenomena.

### **2.2. NFR and river health**

The physical structure of river habitats is predominantly shaped by physical processes, emphasizing water flow and sediment transportation. Various geomorphic characteristics, including sediment size and heterogeneity, river channel and floodplain morphology, and

other relevant geomorphic attributes, determine riverine habitats' nature. For some flow regimes, the flow associated with channel formation differs from the flow that forms the floodplains. This phenomenon is particularly notable in rivers characterized by a broad spectrum of flood flows, where the floodplain can display significant deposits, such as boulder berms adjacent to the channel or other morphological features that are remnants of infrequent but high-magnitude floods (Poff et al., 1997; Miller, 1990). Therefore, the habitat conditions found within river channels and adjacent floodplains vary based on the flow pattern of the river and the type and availability of sediments. As a result, distinctive habitat features such as river bars and riffle-pool sequences are created and maintained by a wide range of flows (Zeiringer et al., 2018). Furthermore, over decades, a river can offer multiple habitat types that include ephemeral, seasonal, and persistent habitats, with flow patterns ranging from free-flowing to standing to completely dry. This habitat diversity of both in-channel and floodplain allowed the evolution of species that capitalize on the habitat mosaic created and sustained by hydrological variability (Sofi et al., 2020). Thus, for many species, a range of habitats is required during their life cycle, which can be maintained by the NFR (Greenberg et al., 1996). Furthermore, the spatial and temporal habitat dynamics are primarily controlled by the flow regime, which, plays role in these species' success, distribution, and abundance (Stanford et al., 1996; Poff et al., 1995). As such, anthropogenic alterations to the NFR disrupt the natural hydrologic variation, leading to alterations in habitat dynamics and placing a strain on the native biota's capacity to adapt.

### **2.3. The Ecological Impacts of Altered Flow Regimes**

The increasing global water demand has led to a contentious situation where the utilization of rivers as both water and energy resources is in direct conflict with the imperative to preserve the ecological status of rivers. The construction of water resource facilities, such as impoundments, diversion weirs, inter-basin water transfers, run-of-river abstraction, and aquifer exploitation, has primarily been undertaken for purposes such as irrigated agriculture, hydropower generation, industry, and domestic supply (Rosenberg et al., 2000). This construction has caused unprecedented disturbances to riverine ecosystems, predominantly attributable to altering the natural hydrological patterns (Rosenberg et al., 2000). Furthermore, with approximately two-thirds of the world's long rivers having been dammed for electricity generation (Grill et al., 2019) and the ever-present risk of further global damming in the future, the river biota that has evolved in

response to NFRs are endangered (Zarfl et al., 2015). Furthermore, this interference with natural hydrological processes disrupts the dynamic equilibrium between water and sediment transport in unaltered rivers (Poff et al., 1997). Thus, altering the flow regime by human activities disturbs the established pattern of natural hydrological variation, exerting stress and degrading physical habitats in streams (Linares et al., 2018). Consequently, this degradation leads to unfavorable impacts on the stream's health, causing the river channel to function unnaturally and irregularly (Brookes et al., 1996). Poff and Zimmerman (2010) conducted a comprehensive review and reported that most of the published research shows adverse ecological consequences due to altering various flow components. Only in a limited number of studies, the ecological response metrics increase, suggesting alterations in an ecological structure, such, as an increase in non-native species or non-woody vegetation on dewatered floodplains. The following sections explore the ecological consequences of altered flow regimes on river ecosystems by four ecological response principles established by Bunn et al., 2002.

### **2.3.1. Flow determines physical habitats and biotic composition**

The shape and dimensions of river channels, the spatial arrangement of riffle and pool habitats, and the stability of the substrate are predominantly governed by the interplay between the flow regime and the landscape's underlying geological and topographic characteristics. Additionally, the physical characteristics of the river, such as sediment size and heterogeneity, channel morphology, floodplain, and other geomorphic attributes, dictate the nature of the riverine habitat. Alterations to the natural hydrological patterns can disrupt the balance between water flow and sediment transport within unobstructed rivers (Dunne et al., 1978). These altered flow regimes can lead to a decline in the health and resilience of river communities (Strona et al., 2016). This phenomenon is observed even at fine spatial scales, where minor variations in flow and near-bed velocities can influence the occurrence and population size of specific flora and fauna species (Bunn et al., 2002).

#### **2.3.1.1. Flow impact on habitat**

Altering the NFR of rivers by impounding both low and high flows through the construction of dams impacts hydrological processes, water quality, and riverine biota (Gillespie et al., 2020; Nilsson et al., 2000; Kingsford, 2000; Poff et al., 1997). Dams also disrupt sediment transportation, coarsening the streambed and reducing the availability

of habitats for aquatic species living in or using interstitial spaces (Chien, 1985). Dams also cause a decrease in the magnitude and frequency of high flows, resulting in fine sediment accumulation, sealing in gravel, channel stabilization, and narrowing (Zeiringer et al., 2018). Furthermore, levees and channelization reduce overbank flows, thereby reducing floodplain deposition and the constriction of river channels. The reduced deposition and channel mobility limits the maintenance of species diversity in floodplains and the formation of secondary channels (Sofi et al., 2020). On the other hand, prolonged periods of low water flow have a direct association with reduced aquatic habitat due to the decrease in the volume, area, and depth of the river. Such low-flow conditions limit the availability of food, resulting in a reduction of the feeding niche within the river channel (Brush et al., 2015). Alterations during low-flow periods can greatly affect the spatial characteristics of fast-flowing environments. For example, a study in West Virginia, USA, attributed the decline of riffle habitat areas to a decrease in discharge (Hakala et al., 2004). As the period of low flow extends, the transport of sediments and silts decreases and benthic sedimentation increase (James et al., 2008). Consequently, the mesohabitat heterogeneity decreases, while riffles, pools, and runs become more homogeneous (Sofi et al., 2020). Moreover, regulating flow and dam construction can fragment rivers, leading to the loss of natural connectivity of river ecosystems (Nilsson et al., 2000). The loss of connectivity between river ecosystems impacts community stability (Allesina et al., 2012), robustness (Strona et al., 2016), and resistance to invasion (Baiser et al., 2010).

#### **2.3.1.2. Flow impact on aquatic plants**

As human water usage intensifies and anthropogenic activities increasingly alter river systems, there is a growing need to understand and predict the potential impacts of flow alteration on the vegetation dynamics within these ecosystems. The assemblage structure of aquatic macrophytes is primarily determined by flow-related factors such as flow extremes, flow regime, hydraulics, substrate composition, and stability (Bunn et al., 2002). The structure of plant assemblages is subject to spatial and temporal variation, which is influenced by flooding and scouring, desiccation, substrate stability, as well as localized variations in water velocity, turbulence, and shear stress (Chambers et al., 1991; Biggs, 1996). Furthermore, the balance between biomass gain and loss processes depends on the flow regime, specifically magnitude, variability, and discharge frequency (Goldenberg et al., 2022). Biomass gains, such as the accumulation of algae and

macrophytes, may occur during periods of increased flow stability and are even more pronounced in altered rivers due to the steady flows associated with damming or water abstraction (Ponsatí et al., 2015; Smolar-Žvanut et al., 2014). In contrast, processes leading to biomass loss are induced by elevated shear stress during high-flow events, which can cause breakage and dislodgement of biofilm mats and plants (Franklin et al., 2008). Nonetheless, some studies report an increase and a decrease in biomass benthic algal communities following significant floods (Goldenberg et al., 2022). Similarly, while there are instances of macrophyte mass growth in certain rivers with increased winter discharges, it is not a universal occurrence (Johansen et al., 2000). However, in line with the “Intermediate Disturbance Hypothesis”, some authors have proposed that a moderate flow disturbance would result in greater species richness (Riis et al., 2008). Consequently, rivers with altered flow regimes may harbor communities with reduced species richness, as such alterations can either augment or diminish the natural dynamism of rivers (Goldenberg et al., 2022). Furthermore, dams trap sediments that move downstream, causing severe consequences, such as the erosion of fine sediment in the lower course of the river (Zeiringer et al., 2018). The flow regulation and disruption of sediment regimes significantly impact riverine riparian ecosystems. These impacts include shifts in riparian plant community composition (Merritt et al., 2010), reduction in native riparian forests (Stromberg et al., 2007), and increased non-native plant species along rivers (Ringold et al., 2008). Moreover, changes in the altitude of landforms significantly impact the annual inundation and moisture patterns, thereby influencing the distribution of vegetation (Sparks, 1992).

### **2.3.1.3. Flow impact on aquatic invertebrates**

Streams and rivers exhibit a rich array of benthic invertebrates, which demonstrate a high degree of biodiversity. These invertebrates also serve a central role in the functioning of river ecosystems. Furthermore, they display intriguing adaptations to the flow characteristics of their aquatic habitats, including life history, nutrition, respiration, and behavioral and morphological traits (Bellard et al., 2012; Lytle et al., 2004). Bunn et al. (2002) reported that physical disturbances from floods and droughts are major factors affecting the spatial and temporal dynamics of benthic communities in streams. Streams with unstable substrates tend to have low species diversity, where the present biota is often adapted to frequently disturbed environments. Macroinvertebrates are particularly vulnerable to rapid changes in flow, and regulated river reaches below hydroelectric dams

with irregular flow patterns typically support communities with low macroinvertebrate diversity. Sudden increases in flow can lead to downstream drift, causing significant losses of benthic biota. High flow water releases below hydropower dams can be highly selective, resulting in the under-representation of invertebrates that cannot tolerate high velocities in downstream areas. Additionally, among the flow parameters that Kakouei et al. (2017) considered in their study, "Duration of high flow events" exhibited the strongest correlation with the density of individual taxa, while the "rate of change of average event" followed. Specifically, 41% and 38% of the taxa demonstrated a peak in their probability of occurrence at ranges of these parameters, respectively. According to a study conducted by Linares et al. (2018), the diversity of benthic macroinvertebrates was reduced within reservoirs of dams. However, an increase in diversity was observed immediately downstream of the dam.

#### **2.3.1.4. Flow impact on fish**

The primary determinant of the structure of fish communities and the spatial patterns of species invasions in rivers is the flow regime, as indicated by both theoretical models and empirical evidence (Gido et al., 2013). Alterations to the flow regimes of lotic systems, whether caused by natural or anthropogenic factors, exert significant influences on the abundance and recruitment of lotic organisms by modifying the quality and availability of spawning habitats, affecting the availability of food resources, and constraining dispersal (Bunn et al., 2002; Poff et al., 1997). In specific systems, and depending on timing, magnitude, and duration of flows, fish populations may benefit from natural flow regimes that exhibit high peak flows. This is due to the ability of high flows to provide connectivity to floodplain habitats, cleanse spawning habitats of fine sediments, and stimulate ecosystem productivity (King et al., 2009; Poff et al., 1997; Lytle et al., 2004).

Mims et al. (2012) investigated the associations between specific flow attributes and the relative prevalence of fish species that represent the three life-history strategies defined by Winemiller et al. (1992) across the United States. Their findings revealed that fish species with opportunistic strategy, characterized by maturity at a young age and rapid turnover, were more commonly found in streams with high flow variability than those with predictable seasonal flows. In contrast, periodic strategists, characterized by maturity at considerable age and high fecundity, were positively linked to high flow seasonality and negatively linked to flow variability. Furthermore, equilibrium strategists, characterized by increased parental investment in offspring, were negatively associated

with flow variability and positively associated with predictability. Rolls et al. (2014) suggested that the impacts of flow regime alterations on freshwater fish follow a four-stage sequence that depends on the magnitude and duration of flow alterations. The first stage is the reduction of population size in regulated rivers compared to unregulated rivers, caused by flow alteration. For instance, altering flow components changes the dynamics between predators and prey in smaller pools (Abebe et al., 2020). The second stage is marked by the density increase of invasive species, followed by a reduction in the density of native species during the third stage. This succession of events is responsible for altering the composition of fish assemblages during stages two and three. Finally, the fourth stage is species extinction, leading to decreasing species diversity.

### **2.3.2. Life history strategies of aquatic species**

Throughout history, river ecosystems and biota have evolved to withstand naturally changing flow regimes and the cyclical patterns of floods and droughts (Lytle et al., 2004; Lytle et al., 2017). Many studies have demonstrated that the interplay between extrinsic drivers (e.g., flow conditions) and intrinsic characteristics of species play a crucial role in determining how species respond to different dimensions of the flow regime (Mims et al., 2012; Gido et al., 2013). The impact of river flow on the recruitment and growth of aquatic plants is well studied. Alterations in the intensity and frequency of water level fluctuations and disturbance can impact both seedling survival and the rate of plant growth (Bunn et al., 2002). This is attributed to the limited tolerance and inability of aquatic plants to regenerate under altered flows (Rea et al., 1994). However, life history traits possessed by some aquatic plants may enhance their performance in altered flows, such as high seed dispersal, ability to germinate while submerged, rapid growth rate, and reproduction (Bunn et al., 2002).

Flow regimes have a substantial impact on morphological, physiological, and behavioral characteristics of organisms (Lytle et al., 2004). As a result, from both ecological and evolutionary perspectives the functional composition of stream communities is impacted by flow components such as, magnitude, predictability, and intermittency (Bunn et al., 2002; Craven et al., 2010). Stream flow alterations caused by human activities will have dissimilar impact on native and non-native fish populations, particularly when these species possess ecological traits that have evolved under different flow regimes (Gido et al., 2013). Therefore, the natural flow paradigm postulates that native species perform better in NFRs, while non-native species may be favored by altered flow regimes. This

paradigm was supported by the findings of Kiernan et al. (2012), which showed that restoring the natural flow and thermal regimes supported native species and repressed non-natives. Nonetheless, a study in North America by Gido et al. (2012) showed that the impacts of altered flow on fish species might be multifaceted. The findings showed that opportunistic non-native fish species were favored by unnatural stable low flows during summer, presumably due to the stable spawning areas and increased water temperature associated with the altered flow. On the other hand, equilibrium native strategists benefited from the lower peak discharge, higher base flow, and reduced fluctuations during the spring discharge. Thus, the nativity of species within an ecosystem, along with their life-history traits, could be a predictor to the response of species to alterations in the natural flow components.

### **2.3.3. Flow and connectivity maintenance**

The inter-system transfer of energy and nutrients, known as subsidies, constitutes a fundamental characteristic of landscapes that governs the productivity and community dynamics of the receiving systems (Giling et al., 2015). Wohl et al. (2017) defined connectivity as the ability of both matter and organisms to transit between spatially defined units within a natural system. The degree of connectivity within natural systems commonly exhibits spatial and temporal variability. However, anthropogenic activities pose a threat to the integrity of these exchanges. Hydrologic alterations caused by water extraction, damming, or altered precipitation patterns can significantly alter the magnitude and temporal patterns of flow extremes, resulting in the disruption of longitudinal, lateral, and vertical connectivity in river networks.

#### **2.3.3.1. Lateral connectivity**

Rivers receive substantial subsidies from neighboring ecosystems since they are at low points in the landscape and possess high edge ratio with the terrestrial system (Richardson et al., 2010). In stream ecosystems, the accumulation, availability, and transport of organic matter subsidies derived from terrestrial sources are governed by the magnitude and timing of high and low flow extremes (Vidon et al., 2010). These factors play a crucial role in supporting the food webs and productivity of the stream ecosystem. Furthermore, flood-induced lateral expansion of floodplain habitats plays a crucial role in providing critical spawning, nursery, and foraging grounds for numerous fish species as well as various other vertebrate taxa (Clarke et al., 2008). The spatiotemporal characteristics of

river inundation events are critical factors in determining the accessibility of fish to nursery habitats and food resources, as well as their susceptibility to being entrapped in isolated floodplain water bodies, or alternatively, being released back into the main river system (Bunn et al., 2002). Thus, the temporal and spatial characteristics of hydrological connectivity between floodplain water bodies and the main river channel during periods of high flow, the frequency and duration of periodic isolation events, and the proximity of these habitats to the river channel are pivotal in shaping the taxonomic and ecological composition of fish assemblages (Ward et al., 1995).

The regulation of flow regimes via dam construction, frequently exacerbated by additional physical modifications such as channelization and levee construction, generally leads to decreased lateral connectivity within floodplain river systems (Ward et al., 1995). Additionally, water storage practices lead to alterations in the natural hydrological peaks. This impedes downstream floodplain inundation, which can result in the unavailability of these areas for seasonal fish spawning and feeding. The resultant exclusion from these habitats can decrease production levels of species that rely heavily on these shoreline habitats, leading to a shift towards those species with greater flexibility in habitat use (Clarke et al., 2008). Consequently, fish communities may become less diverse and productive. Furthermore, the intermittent inundation of floodplain areas through non-regular releases of water from reservoirs can enable fish to utilize these areas for feeding and as shelter from high-flow conditions. Abrupt flow reductions can subsequently lead to fish being stranded in these areas (Nagrodski et al., 2012). Stranding is a commonly observed lateral impact, that often occurs during specific flow management operations such as hydropeaking, and as such, is addressed in a separate section (see sections below).

### **2.3.3.2. Longitudinal connectivity**

The installation of hydraulic flow control structures causes evident impacts on the longitudinal connectivity of rivers, with river corridor fragmentation being one of them. This fragmentation impedes the free movement of fish and other materials (Hanrahan et al., 2004). The ability of aquatic organisms to move freely throughout the stream network is crucial for ensuring the sustainability of their populations (Bunn et al., 2002). However, flow management scenarios can impact the movement of fish in both upstream and downstream directions. The energetic cost of migration is substantial, and unfavorable environmental conditions, such as elevated temperatures and high discharge, often

impede migration (Clarke et al., 2008). The flow structures acting as significant barriers to migration, can negatively impact the timing of arrival at spawning grounds and, consequently, spawning success. Hydroelectric dams are recognized as a cause of delays in the upriver migration of migratory species because fish are attracted to high water velocities and discharge, conditions typically found in the tailraces of hydroelectric plants (Thorstad et al., 2005). Therefore, when fish encounter channels with differing discharge, they are drawn to the channel with the highest velocity, which is frequently the tailrace rather than the opening to a fish passage structure, resulting in a "false attraction" to the tailrace area and migration delays (Clarke et al., 2008). This delay can lead to injuries, increased exposure to predators, a greater risk of disease, and environmental stress. Fish may also exhibit wandering behavior in search of a sustained directional flow or reverse migration direction entirely, further delaying migration (Thorstad et al., 2005). The management of flow can also induce modifications in migration patterns during downstream fish movement. The travel time of downstream migration is primarily related to water velocity, given that downstream migrants travel passively, particularly during periods of high flow that are linked to snowmelt and runoff (Clarke et al., 2008).

#### **2.3.3.3. Vertical connectivity**

Groundwater represents a critical vertical component in river systems, and the zone where groundwater intermixes with surface waters is denoted as the hyporheic zone (Jones et al., 1999). The spatial extent of the hyporheic zone is subject to variability among rivers and depends on diverse physical factors, including discharge and the permeability of adjacent soils. In rivers the water exchange in the hyporheic zone is regulated by dissimilarities in hydraulic head (Clarke et al., 2008). As the variation in discharge and hydraulic head is closely associated with changes in the hydrograph (Malcolm et al. 2004), any form of management intervention that alters the natural hydrograph can potentially influence the discharge and quality of hyporheic water (Calles, 2005). Therefore, modifications to the hyporheic zone's quality could result in unfavorable effects on the ecological services that it delivers, such as incubation habitat for salmonoid species (Malcolm et al. 2004).

#### **2.4. Environmental Flow (E-flows)**

E-flows can be described as the quantity of water that remains within an aquatic ecosystem or is introduced into it, with the explicit aim of regulating the ecosystem's state

(Zeiringer et al., 2018). The field of environmental flows is founded upon the NFR paradigm, which emphasizes the importance of temporal fluctuations in stream flow in sustaining high levels of native biodiversity. This notion has been developed based on the insights gained by riverine ecologists over time, which prompted E-flows to expand in scope and impact. E-flows are now recognized as an influential driver of water resources policy at local, regional, and national levels (Acreman et al., 2014). The prevailing objective of present-day E-flow management is to counteract changes to specific components of the NFR with the goal of reinstating specific ecological traits (Poff, 2018). However, new challenges are emerging that necessitate revisiting the fundamental principles that underpin current E-flow practices. Considering the NFR paradigm (see section 2), river management through E-flows should aim to reinstate a variety of flow conditions, as opposed to a fixed, unchanging flow state such as the minimum flow. The objective of the following sections is to assess the challenges that arise from non-stationarity in hydrology, ecology, and climate and to investigate possible approaches for E-flows science to shift from relatively static to more dynamic and time varying characterizations of flow.

#### **2.4.1. Hydrologic and ecological non-stationarity**

In the past, water resources management was primarily based on the assumption that the means and variances of hydro-climatic processes that underlie hydrologic regimes were generally consistent throughout the Holocene epoch i.e., last 10 to 11 thousand years (Poff, 2018; Milly et al., 2008). This assumption of hydro-climatic stationarity enabled the notion that within undisturbed catchments, the precipitation and runoff processes occur within a discernible range of variability and can be described using long-term averages. Thus, to establish a “reference” case for e-flow practices, water resource managers utilized this assumption of discernible processes to define ecologically significant components of the flow regime (see section 2). However, over the past decade it has become evident that the Earth's climate system is experiencing changes. As a result, we are currently transitioning into the Anthropocene epoch, which is characterized by swift global warming, which, when combined with the global transformation of land surfaces, is responsible for shifts in both the means and variances of temperature and precipitation patterns (Milly et al., 2008). The new epoch leads to non-stationarity in hydrological baselines, which poses significant challenges for the field of E-flow science. This phenomenon results in altered long-term regime averages and has the potential to

bring about regime transformations, possibly leading to novel regimes when compared to historical norms (Laizé et al., 2014; Williams et al., 2007). Therefore, the utilization of past hydrologic time series as a reference condition for the implementation of E-flow management practices is subject to considerable doubt (Acreman et al., 2014; Kopf et al., 2015).

Another formidable aspect of the Anthropocene epoch is the acknowledgement that ecological systems exhibit non-stationarity. The traditional assumption was that local ecological disequilibrium is stabilized at the regional-scale. But human activities have extensively affected ecosystems at all levels, rendering this assumption obsolete (Poff, 2018). The non-stationarity of ecological processes in aquatic ecosystems has been induced by the legacy effects of human activities and the proliferation of non-native species that interfere with biotic interactions and undermine the validity of "reference" conditions in freshwater systems (Thompson et al., 2018; Hobbs et al., 2006; Kopf et al., 2015). Furthermore, the non-stationarity of ecological systems restricts the applicability of past conditions as a reference for the establishment of restoration endpoints (Hobbs et al., 2009).

#### **2.4.2. From static to dynamic environmental flows**

Considering the previous section, the non-stationarity of hydrologic, climatic, and ecological systems requires a shift from static to dynamic hydro-ecological modelling. E-flows have extensively focused on flow regime-based approaches, characterized by employing metrics that quantify long-term regime averages, such as the mean peak, low flow magnitudes, or the mean timing of these events. Such metrics enables the establishment of hydro-ecological relationships that provides a perspective to species evolutionary history, adaptation strategies, and flow alteration responses (see section 2). On the other hand, insufficient recognition has been given to the significance of short-term flow variation that transpires in "ecological time" in the implementation of E-flows (Poff et al., 2018). Although individual flow occurrences contribute to the long-term average regime, they frequently serve as crucial ecological roles, thus determining local dynamics over shorter, management-relevant time scales. Specifically, individual hydrologic extremes, such as high magnitude flows or prolonged droughts, can have ecological ramifications that are unanticipated under the long-term average for those flow components. As a result, populations may be susceptible to local extirpation due to individual events or sequences of extreme events. Therefore, in numerous instances,

ecological responses cannot be accurately predicted through reliance solely on regime-averaged metrics (i.e., static metrics) due to the non-stationary nature of hydro-climatic and ecological conditions. Poff et al., 2018 argued that there is a notable omission from the present E-flow science discussion, that being the lack of acknowledgement that hydro-ecological relationships can be characterized at multiple space-time scales of hydrological and ecological grain. This gap in understanding may be addressed, in part, by adopting the recommendations of King et al. (2015) to improve the monitoring of ecological responses to E-flows. Furthermore, it has been commonly assumed that flow represents the primary variable, and other environmental factors, such as temperature, sediment, and hydraulics, are of secondary importance. However, it is well established within the scientific community that while flow plays a vital role, it is not always a sufficient factor in regulating aquatic and riparian species and ecosystems. Additionally, ecological responses to hydrologic alterations alone exhibit a high degree of variability and may possess limited general transferability (Poff et al., 2010). However, by stratifying data based on river types and carefully selecting ecological metrics, transferability can be enhanced (Chen et al., 2018). Therefore, incorporating such factors into E-flows applications can lead to improved hydro-ecological predictions.

### **3. HYDROPEAKING IMPACTS AND MITIGATION**

#### **3.1. Background**

Hydropower plays a substantial role in the European Union's (EU) renewable electricity generation, accounting for more than third of the overall renewable electricity output and 10% of the gross electricity production as of 2018 (Alsaleh et al., 2023). In comparison to other renewable sources such as wind and solar, hydropower exhibits notable efficiency and flexibility in meeting electricity demand, enabling robust fluctuations in production, even at sub-daily time scales. Therefore, hydropower flexibility, European climate neutrality goals, and global demand growth for renewable electricity are all expected to increase the role of hydropower, especially to balance and stabilize the power market. However, the flexibility of hydropower generation leads to the occurrence of hydropeaking, which is defined as an artificial river flow regime resulting from the cyclical release of water caused by the rapid switching between increasing and decreasing power generation in HPPs. These rapid fluctuations observed in sub-daily river flows can be attributed to the responsiveness of hydropower to shifts in the power market (Bieri et al., 2014). Thus, manipulating power demand can indirectly intensify hydropeaking

regimes in rivers, leading to artificial flow fluctuations (Ashraf et al., 2022). Such fluctuations, posing a significant risk to aquatic ecology, have the potential to induce ecological stresses and compromise the ecological sustainability of rivers and their associated floodplains. Furthermore, these alterations can have adverse effects on the physical habitats of streams, which play a vital role in shaping the biotic and abiotic composition as well as the properties of rivers.

### **3.2. Hydropeaking impacts**

The practice of hydropeaking results in noteworthy fluctuations in flow patterns, exerting a substantial impact on river ecosystems. In comparison to seasonal or daily flow variations, the alterations in flow patterns induced by hydropeaking are more pronounced at temporal scales. Additionally, hydropeaking releases often exhibit significantly higher magnitudes than natural flows, leading to significant modifications in water depth and velocity. Consequently, water quality, flow velocity, water temperature, and depth, are adversely impacted. In turn the survival and reproduction of fish and other aquatic organisms is negatively influenced. Furthermore, recreational use of river systems can also be impacted. The occurrence of downstream flow regime fluctuations, potentially resulting in up to five hydropeaking events per day, is contingent upon the extent of HPPs contribution to meeting the demands of the power market (Schmutz et al., 2018).

#### **3.2.1. Sediment transport and channel morphology**

A primary constraint impeding the natural functioning of ecosystems is the elevated instability of channel habitats, whereby hydropeaking alters the hydraulic flow, sedimentary composition of riverbeds, sediment movement, and habitat availability (Vericat et al., 2020). In fluvial systems, sediment distribution is critical in determining the mobility of particles of varying sizes. Fluvial sediments vary in both the cross-sectional and vertical dimensions of the channel. The degree of mobility of river-bed particles is primarily controlled by the presence of a coarse surface layer i.e., armour layer. In addition, bed structuring affects particle hiding and protrusion, and finer particles may require a higher threshold than larger particles to become mobile. Therefore, accurately characterizing sediment mobility of a given river reach requires an understanding of the frequency at which the flow exceeds the mobility threshold and the magnitude of the excess energy of the flow.

The existence of small sediment patches in riverbeds is important for the proper functioning of rivers and the dynamics of the ecosystem. Batalla et al. (2010) reported that the depletion of such patches occurs when there is no replenishment from upstream or when flow conditions fail to destabilize the protective layer. Additionally, the supply of sediment also influences bed structure and mobility. In rivers impacted by hydropeaking, where water flow is artificially increased without a concurrent supply of sediment from upstream, the process of partial bed mobility resulting from the entrainment of sediment from patches is common. This condition creates a sedimentary deficit that can adversely affect various processes, such as fish spawning and invertebrate refuge. Vericat et al. (2020) suggested that sediment dynamics and availability differ between the control reach they studied and downstream reach's that are affected by daily hydropeaking. The study added that the impact of hydropeaking was grain size specific, such as sand and fine gravel are consistently entrained, transported, and depleted, while boulders remain stationary. This might result in the depletion of fine sediments from patches and a gradual lack of other ecologically significant grain sizes.

### **3.2.2. Ramping rate**

The rate at which water level increases or decreases is defined as up-ramping and down-ramping rate respectively. Augmented ramping rates can either increase or decrease the duration which flow change occurs. Compelling evidence suggests significant correlation between the ramping rate and the responses of stream organisms. High up-ramping rate reduces the available time for aquatic organisms, such as macroinvertebrates, to seek shelter, resulting in a substantial increase in their drift rates (Imbert et al., 2000). Moreover, exceeding certain up-ramping thresholds can trigger fish drift, which is dependent on the species and life stage of the fish. Auer et al. (2017) demonstrated that reducing up-ramping rates from 3 to 0.5 cm/min leads to a decrease in juvenile grayling drifting. However, schmutz et al. (2018) argued that low down-ramping rates might be a greater influence on fish than high up-ramping rates. A rapid decrease in water levels may increase the probability of stranding for organisms, as they may not be able to respond in time. However, the findings of Thompson et al. (2011) suggest that organisms may develop adaptive responses to repetitive hydropeaking. It is important to note that various taxa exhibit distinct durations of exposure to heightened discharge that they can tolerate, surpassing these thresholds results elevated organism drift. Whether taxa are impacted by

hydropeaking is contingent on species-specific traits, such as morphological and behavioral adaptations.

### **3.2.3. Frequency and timing of hydropeaking**

Defining the impact of hydropeaking may be dependent on parameters such as, the frequency and timing of the event. However, the relationship between these parameters and the impact of hydropeaking appears to be complex, as indicated by multiple studies. Therefore, a thorough understanding of these parameters and their interplay is necessary to define the impacts of hydropeaking accurately. Fish stranding may occur naturally and impact only a small proportion of a fish population at any given time. However, the frequent flow fluctuations can lead to cumulative mortalities that may ultimately result in a substantial loss of fish (Young et al., 2011). In contrast, Schmutz et al. (2018) reported several studies revealing a form of temporal adaptation behavior exhibited by fish. One example, when juvenile graylings were exposed to three peak events occurring within a 24-hour period over the course of 21 days, stranding was only observed during the first nine days. However, if flow conditions remain stable for more than 24 hours prior to a peak event, this adaptive response appears to diminish. Furthermore, given that the activity patterns of aquatic organisms vary over the course of a day, the timing of hydropeaking events represents a crucial parameter. Notably, studies have demonstrated that macroinvertebrate drift rates tend to increase at night when these organisms are more actively engaged in feeding behaviors. This nocturnal increase in feeding activity may be driven by predation pressure during the daylight hours.

### **3.2.4. Water temperature**

Human activities and climate change have a significant impact on the thermal characteristics of rivers, resulting in modifications to the composition of aquatic species and alterations in the rates of biogeochemical processes (Caissie, 2006). The phenomenon known as "thermopeaking" refers to the occurrence of rapid and intermittent changes in the thermal regime of a watercourse, specifically caused by hydropeaking (Zolezzi et al., 2011). It is important to note that thermal regimes vary on sub-daily, daily, weekly, and seasonal scales. Thermal variations at these temporal scales can cause adverse impacts on biotic and abiotic compositions in rivers. However, thermal variation at the seasonal scale might be easier to detect and quantify. Additionally, thermal variation at both spatial and temporal scales is of importance to understand and quantify the influence of various

mechanisms governing rivers temperature. This is accomplished by evaluating the respective contributions of heat advection and diffusion processes, as well as the impacts of external sources in the atmosphere and riverbed (Zolezzi et al., 2011).

The release of water for power production reduces water temperature during peak events in the summer, whereas an increase in temperature occurs during peak events in the winter (Schmutz et al., 2018). Frutiger (2004), used a 30-minute interval temperature dataset to examine the thermal impacts of hydropower releases in the Ticino River, located in Switzerland. The researcher identified the occurrence of multiple temperature peaks per day, which he attributed to the releases from the hydropower plant. The study recognized these temperature peaks as an ecological threat to the river. In latter investigations, Carolli et al. (2012) and Bruno et al. (2013) documented an increase in macroinvertebrate drift, characterized by warm and cold thermopeaking, after hydropeaking events. In contrast, findings from a controlled experimental investigation conducted by Schülting et al. (2016) indicated that the combination of hydropeaking and cold thermopeaking exhibited an antagonistic influence on the drift behavior of aquatic macroinvertebrates. The outcomes propose that the responses of macroinvertebrates to cold thermopeaking vary among taxonomic groups, yet generally result in diminished drift for most taxa. The impact of hydropeaking on fish is related to water temperatures and other flow parameters (see previous subsections), with seasonal variations playing a significant role. Specifically, decreased water temperature during the winter season generally leads to reduced activity levels in fish (e.g., Atlantic salmon and brown trout) (Halleraker et al., 2003). At a sub-daily scale, the occurrence of thermopeaking, can also influence fish responses. Since the activity and metabolism of fish are influenced by water temperature, species experiencing a decline in water temperature during a flow fluctuation may exhibit elevated rates of drift and stranding as a result (Schmutz et al., 2018). Increased drift and stranding with decreasing water temperatures was documented for Juvenile Chinook salmon and grayling (Bradford, 1997).

### **3.3. Hydropeaking mitigation**

Hydropeaking mitigation strategies can be categorized into two distinct groups: direct measures and indirect measures. Direct measures aim to alleviate the hydrological impacts of hydropeaking by leveraging operational measures that can be made to HPPs

mode of operation. On the other hand, indirect measures aim to mitigate the ecological impacts of hydropeaking by means of modifying the river's morphology.

### **3.3.1. Potential measures for hydropeaking mitigation**

Modifying the operational mode of HPPs is a mode of direct measures that could be leveraged by plant operators to mitigate the hydrological adverse impacts of hydropeaking. This mode of direct measures would not require capital expenditure but would entail an economic cost on the HPPs profits. By using this measure, the operator would aim to increase the minimum base flow and reduce flow fluctuations. An alternative mode would be a constructional measure that still qualifies to be a direct measure such as the deployment of re-regulation reservoirs. These reservoirs retain water from peak events and smoothly release it back into the river. Another constructional measure would be the construction of side channels to drain flow from hydropeaking events. Constructional measures require initial capital expenditure without imposing operational economic costs. Indirect mitigation measures aim to mitigate the ecological impacts of hydropeaking, through morphological modifications. Such as widening the river channel which reduces the water level changes caused by hydropeaking. Another indirect mitigation measure is the construction of side channels with a stable flow to serve as refugial habitats. Both direct and indirect mitigation strategies possess the capacity to diminish the hydropeaking effects on aquatic organisms. However, in light of the hydrological, ecological, and climatic non-stationarity coupled with the power market demand increase, a versatile and flexible solution is needed. As such, we introduce a novel direct mitigation measure in the following section.

### **3.3.2. Re-regulation reservoirs (RRRs): Conceptual Approach**

RRRs are designed to capture and hold a surplus amount of water from a river's flow and subsequently discharge it at a controlled rate to minimize any short-term fluctuations in the river's flow pattern, as depicted in Figure 1. The RRR accomplishes this by collecting the excess water (red phase in Fig. 1), typically generated during power production up-ramping events at HPPs and releasing it during down-ramping events (blue phase, Fig. 1) by means of two automatic gates that regulate the inflow and outflow of water in and out of the reservoir. The primary objective of the RRR is to restore the natural flow pattern of the regulated river as closely as possible. Figure 1.a is a visual schematic that illustrates how the RRR operates. The opening and closing of the inlet and outlet gates in the RRR

are governed by a re-regulation algorithm developed in this research. If optimal, the RRR has the potential to entirely restore the natural flow pattern of a regulated river and maintain a steady minimum flow to accommodate the environmental needs of the waterway. However, this may necessitate a substantial RRR volume, which may only sometimes be feasible due to economic or land-use restrictions. In cases where the RRR volume is constrained, it should be operated in a way that prioritizes achieving primary objectives before addressing secondary objectives, as circumstances allow. Figure 1.b illustrates a simplified comparison between a natural river regime, regulated river regime, and a re-regulated river regime, while also demonstrating the occurrence of excess water that could be retained in RRRs (represented by red) and the possible release of flow back into the river (represented by blue) at a sub-daily scale.

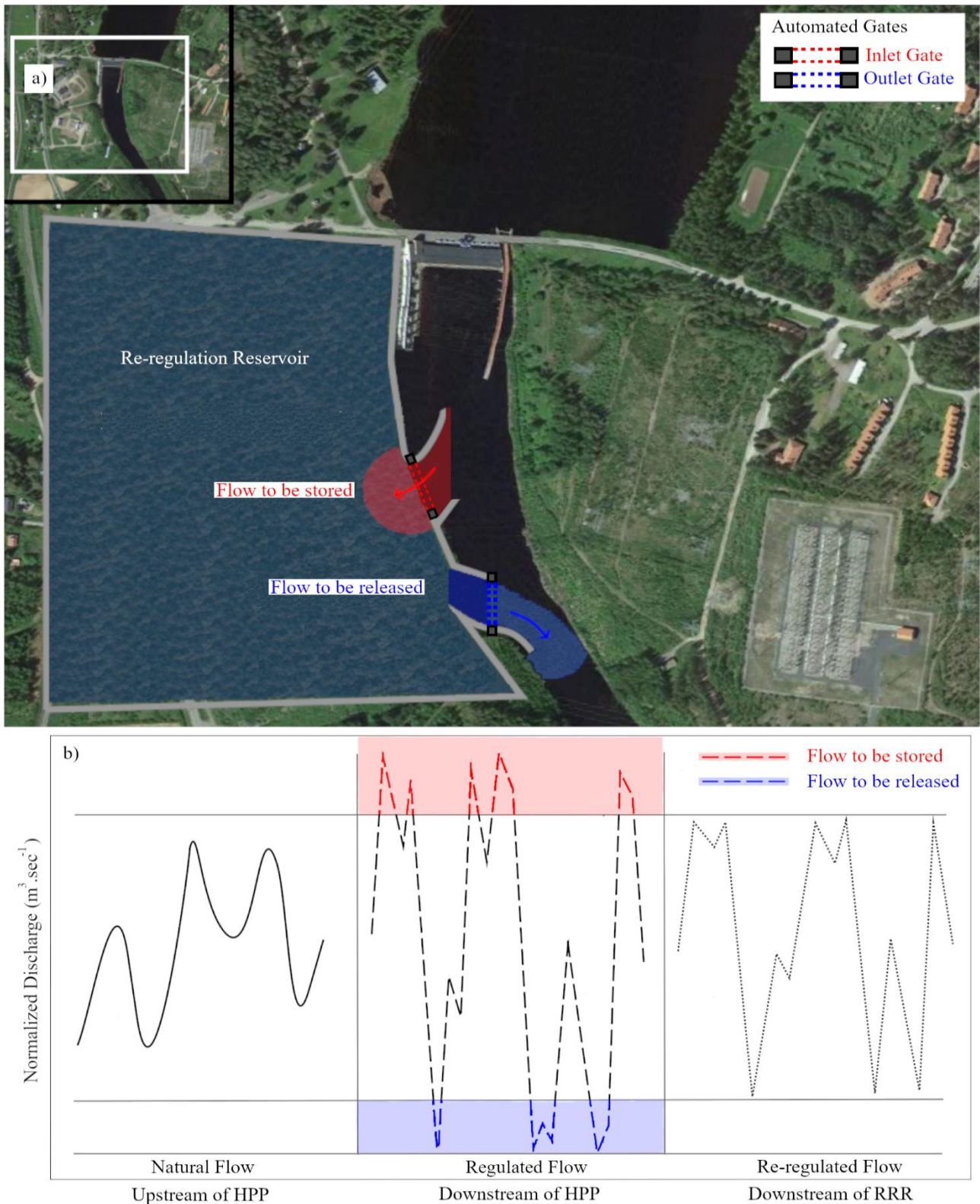


Figure 1. Conceptual depiction of RRR operation, demonstrating the potential storage of peak river flow (referred to as the "red phase") by the RRR and the corresponding flow release back into the river (referred to as the "blue phase"). a) Provides an abstract illustration of an RRR and a river, b) Presents a theoretical hydrograph of the river flow, beginning upstream of the HPP and advancing downstream of the RRR. Google Earth 9.185, (2020) Petäjäsoski, 66° 16' 10" N, 25° 20' 17" E elevation 49m. [Online] Available at: <https://earth.google.com> [Accessed 5 March 2023].

#### 4. MATERIALS AND METHODS: MODEL-BASED DESIGN DEVELOPMENT

Achieving a complete restoration of a river's natural regime necessitates a substantial RRR volume, which may prove unfeasible due to economic or land availability constraints. Nonetheless, attenuating the peak flow, augmenting the minimum flow, and restricting ramp rates can considerably restore the river's natural regime. These objectives may be addressed by implementing RRRs that entail the retention and controlled discharge of flows into the river system (Fig.1). Inadequate (i.e., too slow) water release may lead to a small RRR volume, impeding the accommodation of water from peak flows and up-ramping events. Therefore, for the effective management of RRRs, a model must be developed to determine the timing and quantity of water to be stored or discharged. Once this is achieved, the model can compute the required RRR volume. With this notion in consideration, the theoretical underpinning of the model developed in this study rested upon two primary objectives. The primary objective of the RRR is to curtail the hourly peak flow ( $Q_{\max}$ ) and increase the minimum hourly flow ( $Q_{\min}$ ) that hydropeaking provokes. The secondary aim of the RRR is to diminish the up- and down-ramping rates, alongside increasing the duration during which flow changes transpire. The ramping flow rate ( $\Delta Q(t)$ ) [ $\text{m}^3 \text{ s}^{-1} \text{ min}^{-1}$ ] given in equation (Eq. 1), represents the increase (i.e., up ramping, positive values) or decrease (down ramping, negative values) in the flow over a given time step, where,  $Q(t)$  is the discharge at time  $t$  ( $\text{m}^3 \cdot \text{s}^{-1}$ ),  $Q(t - \Delta t)$  is the discharge at time  $t - \Delta t$  ( $\text{m}^3 \cdot \text{s}^{-1}$ ) and  $\Delta t$  is the time step (min).

$$\Delta Q(t) = \frac{Q(t) - Q(t - \Delta t)}{\Delta t} \quad (\text{Eq. 1})$$

To design the model and analyze how various flow patterns would affect the required RRR volumes, it was necessary to incorporate flow patterns that resembled regulated river regimes which experience frequent hydropeaking. To this end, 24 stations were selected from Finland, covering different rivers, and encompassing several of the most regulated waterways in the country. The selection of the stations was based on the hydropeaking impact classification done by Ashraf et al. (2018), which utilized a hydropeaking indicator and a ramping rate indicator. The selected stations encompassed all three hydropeaking classes i.e., 1, 2, and 3 (Ashraf et al., 2018). Table 1 highlights the main characteristics of the selected stations as well the hydropeaking impact class, while Figure 2 display their spatial distribution. Hourly discharge data for the lower part

of main river channels from 2015 to 2017 was employed. To generate the required flow pattern (hereinafter called scaled flow), characterized by an average discharge of  $1 \text{ m}^3 \cdot \text{s}^{-1}$ , the discharge data was scaled down by dividing the hourly discharge by the average hourly discharge per day (Eq. 2).

$$Q_{scaled}(t) = \frac{Q(t)}{Q_{avg}(d)} \quad (\text{Eq. 2})$$

Where  $Q_{scaled}(t)$  is the scaled discharge at time  $t$  ( $\text{m}^3 \cdot \text{s}^{-1}$ ),  $Q(t)$  is the actual discharge at time  $t$  ( $\text{m}^3 \cdot \text{s}^{-1}$ ), and  $Q_{avg}(d)$  is the average hourly discharge per day ( $\text{m}^3 \cdot \text{s}^{-1}$ ).

After obtaining the scaled flow, a hierarchy of operational objectives with varying thresholds can be established to determine the timing and quantity of water stored or discharged by the RRR. Subsequently, a re-regulation algorithm was formulated to govern the RRR's operations, guided by a set of hierarchical objectives and their corresponding thresholds as follows:

- Priority 1, Reliability and Safety: the inflow and outflow discharges from the RRR would not lead to the overflow or the depletion of the re-regulation basin in the next time-step.
- Priority 2, Peak and Minimum flows: the operation of the re-regulation basin would reduce the peak flow and increase the minimum flow by 10 to 50% (hereinafter referred to as flow adjustment) for all 24 stations. Thus, the peak flow is limited between 50 and 90% $\times$ Qmax and the minimum flow is between 110 and 150% $\times$ Qmin.
- Priority 3, Maximum Up- and Down-ramping rates: The ramping rates (r.r) were selected for each of the 24 stations separately, starting from a r.r threshold below the average r.r and extending to 50% of the maximum r.r (see Table 1). Up-ramping rates are limited to a particular threshold when the flow exceeds the average daily flow. Down-ramping rates are limited to a distinct threshold when the flow is smaller than the average daily flow. To include these r.r thresholds in the scaled flow re-regulation algorithm, they had to be scaled down according to equation (Eq. 3)

$$r.r_{scaled} = \frac{r.r_{Unscaled}}{Q_{avg}(a)} \quad (\text{Eq. 3})$$

Where  $r.r_{scaled}$  is the scaled r.r threshold ( $m^3sec^{-1}min^{-1}$ ),  $r.r_{unscaled}$  is the unscaled r.r threshold, and  $Q_{avg}(a)$  is the annual average discharge ( $m^3sec^{-1}$ ).

- Priority 4, Optimal flow: A flow is restored to a daily average flow whenever possible.
- Priority 5, Optimal up and down ramping rates: Whenever possible, the r.r thresholds are satisfied regardless of if the flow is greater or smaller than the average daily flow.

Given that the optimal flow conditions for diverse ecosystem services may vary, the algorithm utilized a range of thresholds to determine the required volume of the RRR for multiple hydropeaking mitigation scenarios. Thus, the threshold range for flow magnitude and ramping rates was selected to include a range of possible mitigation scenarios. To further broaden the scope of possible mitigation scenarios, thirty-five unique permutations were generated and tested for each of the 24 stations by matching the peak and minimum flow thresholds (i.e., priority 2) with ramp rate thresholds (i.e., priority 3). Hereinafter, the permutations will be referred to as P-z (X%, Y), with X% being the percentage adjusted from  $Q_{max}$  and  $Q_{min}$ , i.e.,  $(100-X) \% \times Q_{max}$  and  $(100+X) \% \times Q_{min}$ , while Y is the r.r threshold, i.e., (Y) for up-ramping and (-Y) for down-ramping, and (z) refers to the station (see Table 1). One example from the permutations is P-a (20%, 1.5) which matches the 20% flow adjustment (i.e.,  $80\% \times Q_{max}$  and  $120\% \times Q_{min}$  threshold) with a ramp rate threshold of  $1.5 m^3s^{-1}min^{-1}$  at Lieksankoski station. Additionally, we demonstrate how the RRR would re-regulate the flow downstream of Juvankoski for permutation P-h (10%, 0.035) by using the re-regulation algorithm.

This model determined the required volume of RRRs downstream of HPPs operating at the rivers mentioned in Table 1. The model can potentially be utilized in other rivers to achieve the above listed priorities. However, the range of thresholds employed by the re-regulation algorithm must be modified to best suit the flow pattern of the investigated river. It is important to note that the model assumes the location of the RRR is immediately downstream of the HPP, thus not accounting for flow velocity or the time required for water to reach the RRR. The model also assumes ideal RRR conditions without considering any water losses that might occur due to evaporation and seepage.

Table 1. Mean discharge (2015-2017), river system, assigned abbreviations (Abb.), and coordinates (TM35FIN) of the selected stations.

Station name	River System	Abb.	Impact classification	Mean discharge (m <sup>3</sup> s <sup>-1</sup> )	Coordinates	
Lieksankoski	Lieksanjoki	a	2	108.32	7026588 N	652025 E
Pamilon	Palojoki	b	3	73.02	6968072 N	674810 E
Leuhunkoski	Leuhunjoki	c	3	23.37	6949767 N	411363 E
Hietamankoski	Suojoki	d	3	32.80	6942209 N	426439 E
Kannuskoski	Kymijoki	e	2	6.53	6758924 N	513225 E
Anjalankosken	Kymijoki	f	2	294.68	6729206 N	490034 E
Peltokoski	Karjaanjoki	g	3	16.25	6672314 N	324236 E
Juvankoski	Tarvasjoki	h	3	5.31	6723054 N	267692 E
Juntola	Tarvasjoki	i	3	7.58	6719220 N	264338 E
Kyröskoski	Pappilanjoki	j	3	25.70	6843083 N	298314 E
Sallilankoski	Loimijoki	k	3	17.18	6773327 N	267734 E
Ryötönkoski	Inhanjoki	m	2	8.44	6934888 N	351480 E
Vatajankoski	Karvianjoki	n	3	10.84	6875136 N	250212 E
Niiles	Jalasjoki	o	3	4.99	6947236 N	267179 E
Kalajärven	Seinäjoki	p	2	3.19	6941625 N	299128 E
Hanhikoski	Kyrönjoki	q	3	41.43	6988348 N	282529 E
Vääräkoski	Jänisjoki	r	2	13.12	6935984 N	347830 E
Kattilakoski	Esse	s	1	17.98	7043283 N	315994 E
Kaitfors	Melakosk	t	3	25.85	7068422 N	321315 E
Porttipahta	Kemijoki	u	3	66.00	7538730 N	489352 E
Kokkosniva	Kitinen	v	3	131.06	7456197 N	513991 E
Seitakorva	Kemijoki	w	3	374.06	7372976 N	515721 E
Taivalkoski	Kemijoki	x	3	741.68	7302096 N	3387909 E
Myllykoski	Kuusinkijoki	y	2	11.76	7339659 N	615246 E

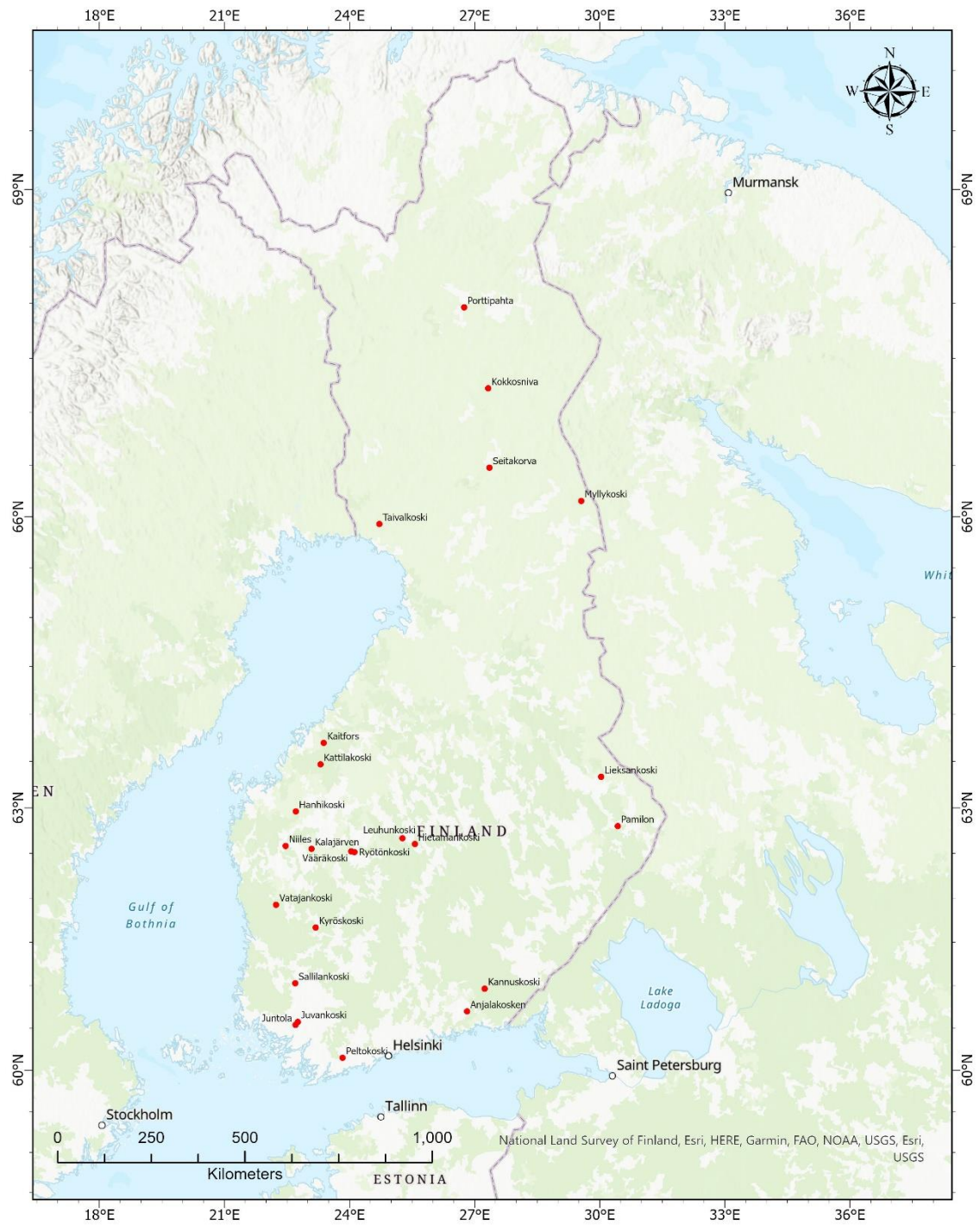


Figure 2. Map of Finland displaying the spatial distribution of the selected stations

## 5. RESULTS

Based on our calculations, we have identified a theoretical potential for using RRRs to regulate hydropeaking. The required RRR volumes for 35 permutations for each of the 24 stations are displayed in Table 3 below. Under the assumption that future flow patterns remain consistent with the scaled flow of the study period, our findings indicate that the required volumes of RRRs outlined in Table 3 are sufficient to maintain compliance with hydropeaking thresholds year-round. The results indicate that, for most of the tested stations at various permutations, the required volume of the RRRs is positively correlated to the thresholds for peak and minimum hourly flows and the ramping rates. Stations that have a positive linear correlation between the required volume of the RRRs and the thresholds are listed in Table 2 below. As the thresholds for these stations became more stringent, the required RRR volume increased. Nonetheless, this trend was not observed at some stations, where several permutations did not have a positive linear relationship between thresholds and required RRR volumes.

Table 2. The relationship between the required RRR volume and flow thresholds for different stations

	Positive linear relationship	Distinct relationship
Stations	(b), (c), (d), (f), (h), (i), (j), (m), (n), (o), (p), (u), (v), (y)	(a), (e), (g), (k), (q), (r), (s), (t), (w), (x)

For station (a), P-a (10%, 5), P-a (10%, 6), and P-a (10%, 7) required a smaller RRR volume than P-a (20%, 5), P-a (20%, 6), and P-a (20%, 7) respectively. The latter permutations have a 20% flow adjustment which is more stringent than the 10% flow adjustment of the former permutations. Several permutations in stations (e), (r), and (t) displayed the same correlation as the station (a) between the required RRR volume and the thresholds when the flow adjustment increased from 10% to 20%. As for station (g), the required RRR volume remained almost constant for all permutations when the flow adjustment threshold increased from 10% to 20%. However, when the flow adjustment increased from 20% to 30%, the required RRR volume decreased for all permutations, to then increase as the flow adjustment threshold was set higher than 30%. Apart from the flow adjustment threshold, the correlation between the ramping rate threshold and the required RRR volume also displayed non-linear relationships. This is evident in the

station (k), where P-k (20%, 0.510) requires a larger RRR volume than P-k (20%, 0.425), P-k (20%, 0.340), P-k (20%, 0.255), P-k (20%, 0.170), and P-k (20%, 0.082) which more stringent ramp rate thresholds. Station (q) displayed a unique correlation with the required RRR volumes decreasing as the ramp rates thresholds became less stringent, and the flow adjustment thresholds became more stringent (10% to 40%). However, when the flow adjustment threshold was set to 50%, the required RRR volume increased substantially. Station (r) also displayed a distinctive correlation, with the required RRR volume for permutations having a ramp rate threshold of 0.007 and 0.014  $\text{m}^3 \cdot \text{s}^{-1} \cdot \text{m}^{-1}$  remaining constant. Additionally, the required RRR volume for the remainder of the permutations in station (r) remained constant when the flow adjustment threshold increased from 10% to 20% and from 30% to 50%. However, the required RRR volume increased when the flow adjustment threshold was elevated from 20% to 30%. For station (x), the required RRR volume for permutations P-x (10%, 3.5, volume: 0.256 million cubic meters (MCM)) and P-x (10%, 4, volume: 0.368 MCM) are larger than the reservoir volume needed for permutations P-x (10%, 2.5, volume: 0.147) and P-x (10%, 3, volume: 0.143) which have more stringent ramp rate threshold. Furthermore, for permutation, P-x (10%, 4), the required RRR volume (0.368 MCM) decreases slightly compared to P-x (20%, 4, volume: 0.262 MCM), which has more stringent peak and minimum flow thresholds. Our theoretical approach demonstrates the relationship between the required RRR volume, daily peak discharge, and ramp rate thresholds for the 24 selected stations in Figure 3. Furthermore, each month's required RRR volume was determined separately at Taivalkoski, Juvankoski, Pamilon, and Kaitfors stations, illustrated in Figures 4, 5, 6, and 7. The results indicate that for most of the investigated permutations at Taivalkoski station, except permutations with 50% flow adjustment, July and August require the most enormous RRR volume to achieve the objectives and priorities in section 4. Whereas, for permutations with a 50% flow adjustment, January is the month that requires the largest reservoir volume. However, it is essential to note that January and February are the months with the largest volume requirement when the ramp rate thresholds exceed 2.5  $\text{m}^3 \cdot \text{s}^{-1} \cdot \text{min}^{-1}$  for permutations with a 10% flow adjustment. On the other hand, for stations Juvankoski, Pamilon, and Kaitfors, September and October require the largest RRR volume for all the permutations. Additionally, the RRR operation for P-h (10%, 0.035) for the scaled flow downstream of Juvankoski is demonstrated in Figure 8.

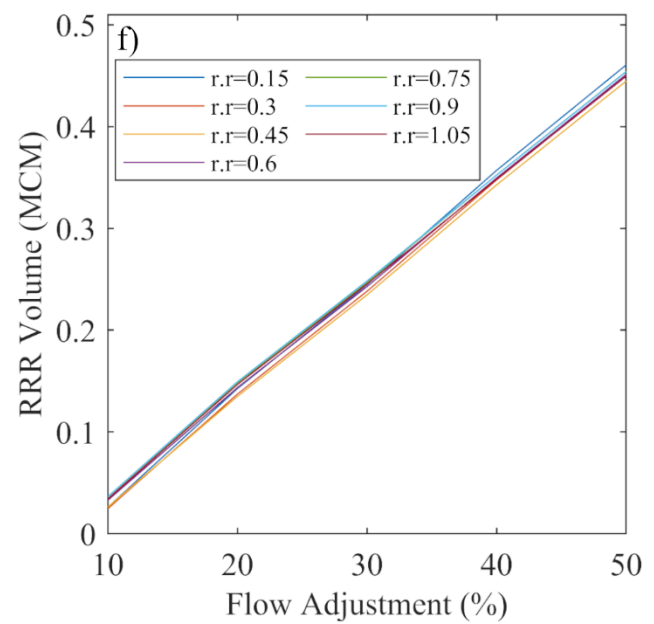
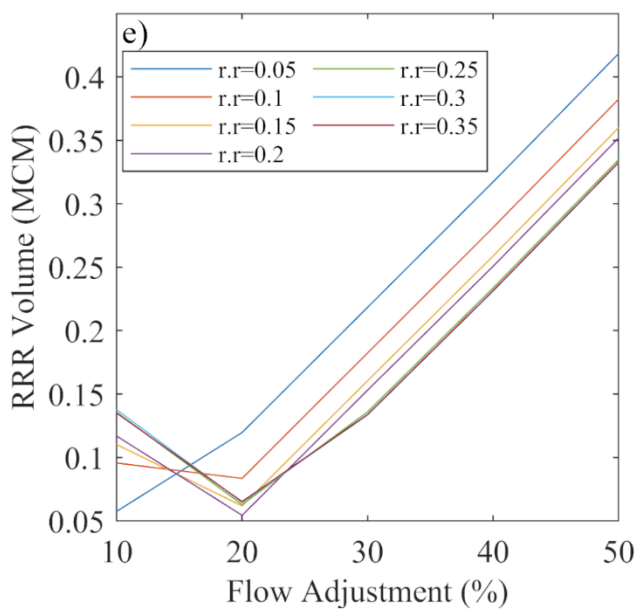
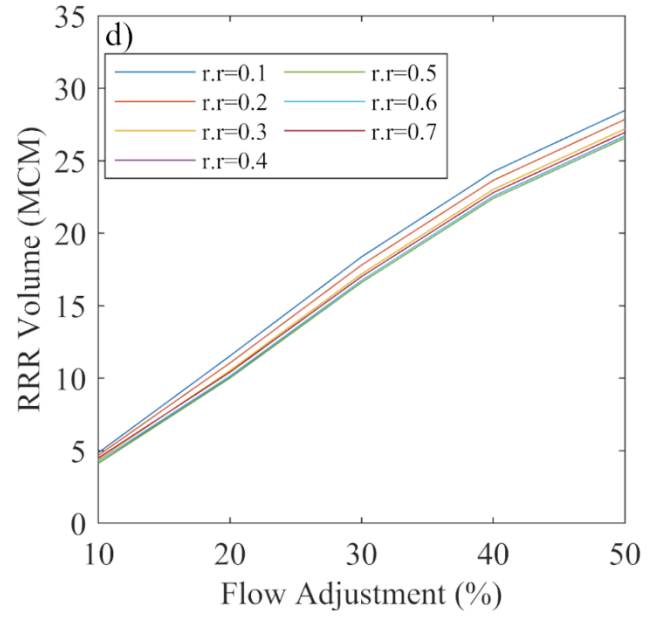
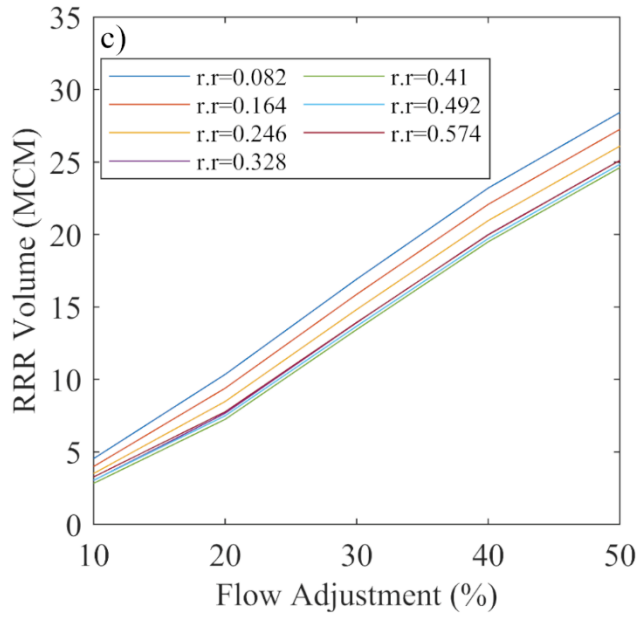
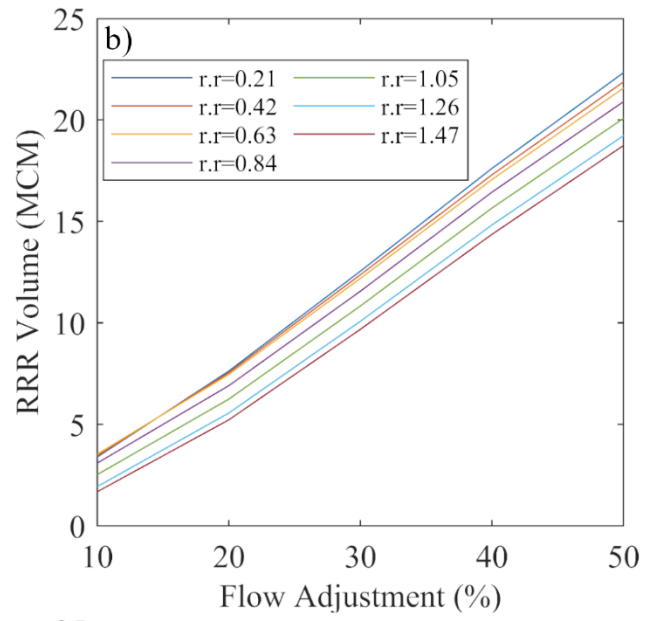
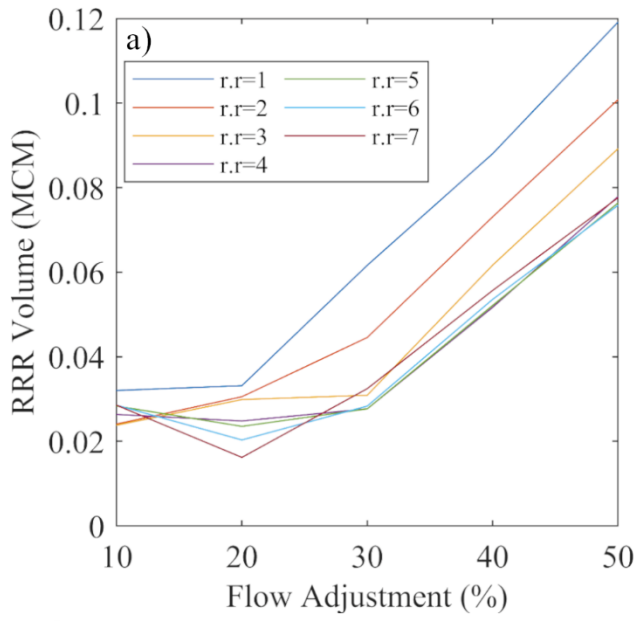
Table 3. The required re-regulation reservoir volumes (MCM) for different stations and permutations (scaled flow from 2015 to 2017).

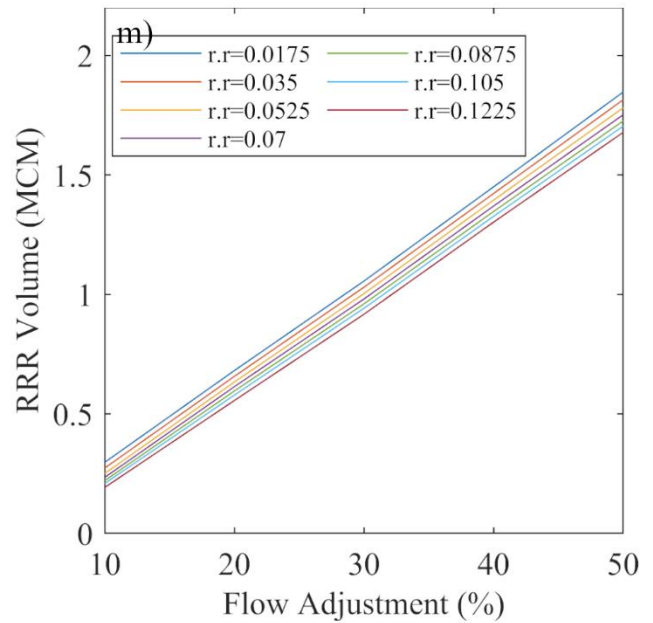
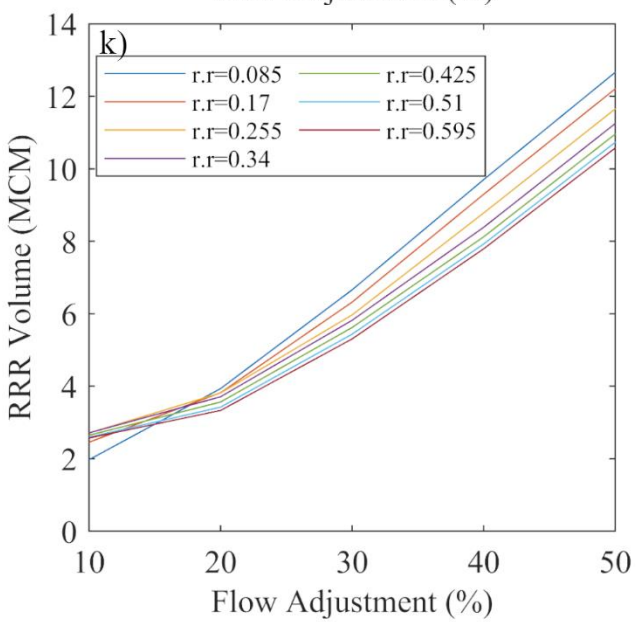
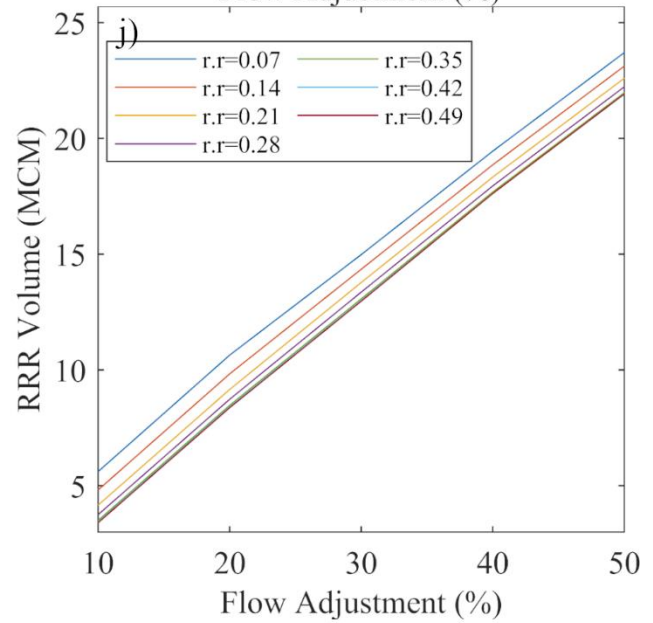
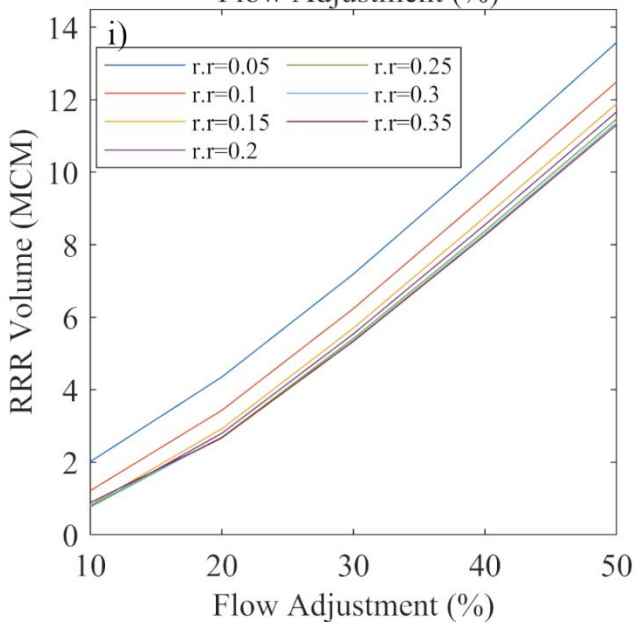
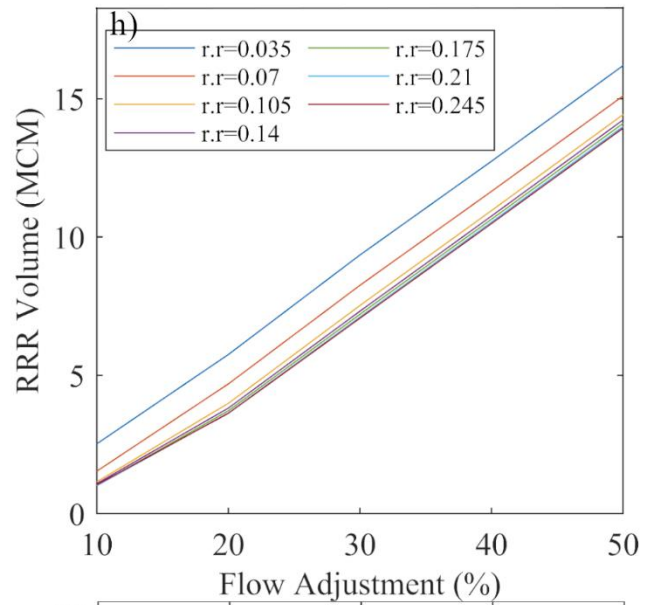
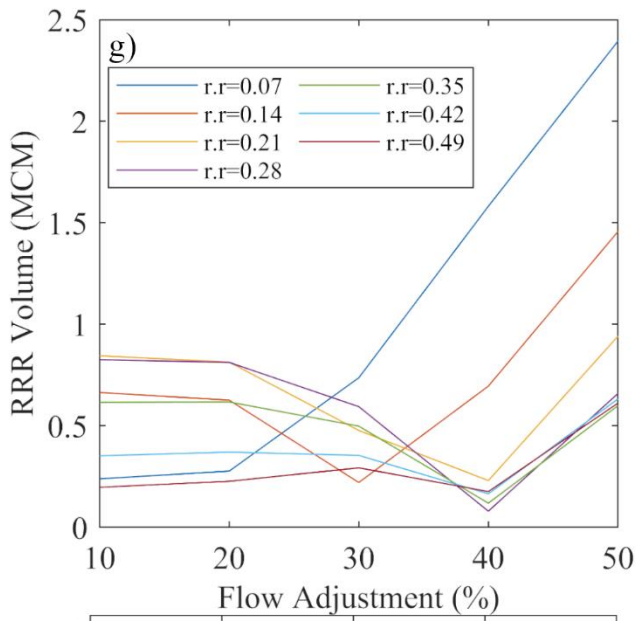
Flow Adj. r.r	a) Lieksankoski							b) Pamilon						
	1	2	3	4	5	6	7	0.21	0.42	0.63	0.84	1.05	1.26	1.47
10 %	0.032	0.024	0.023	0.026	0.028	0.028	0.028	3.392	3.461	3.534	3.095	2.527	1.936	1.683
20 %	0.033	0.030	0.029	0.024	0.023	0.020	0.016	7.632	7.552	7.465	6.909	6.247	5.562	5.226
30 %	0.061	0.044	0.030	0.027	0.027	0.028	0.032	12.553	12.376	12.194	11.565	10.837	10.082	9.687
40 %	0.088	0.073	0.061	0.051	0.052	0.053	0.055	17.617	17.300	17.068	16.447	15.661	14.831	14.368
50 %	0.119	0.100	0.089	0.077	0.076	0.075	0.077	22.334	21.883	21.574	20.908	20.084	19.232	18.748
	c) Leuhunkoski							d) Hietamankoski						
	0.082	0.164	0.246	0.328	0.410	0.492	0.574	0.1	0.2	0.3	0.4	0.5	0.6	0.7
10 %	4.534	3.988	3.506	3.033	2.829	3.040	3.270	4.855	4.677	4.390	4.132	4.115	4.274	4.497
20 %	10.346	9.394	8.477	7.663	7.246	7.490	7.769	11.533	11.047	10.513	10.108	10.003	10.150	10.414
30 %	16.922	15.872	14.827	13.918	13.443	13.677	13.937	18.371	17.805	17.195	16.739	16.603	16.743	17.012
40 %	23.207	22.080	20.961	19.999	19.498	19.720	19.975	24.244	23.650	23.020	22.546	22.396	22.531	22.797
50 %	28.421	27.255	26.098	25.113	24.615	24.845	25.105	28.468	27.848	27.195	26.709	26.553	26.684	26.950
	e) Kannuskoski							f) Anjalankosken						
	0.05	0.1	0.15	0.2	0.25	0.3	0.35	0.150	0.300	0.450	0.600	0.750	0.900	1.050
10 %	0.057	0.096	0.11	0.117	0.136	0.138	0.135	0.026	0.025	0.026	0.033	0.034	0.036	0.034
20 %	0.12	0.083	0.062	0.054	0.063	0.065	0.065	0.143	0.137	0.135	0.144	0.148	0.149	0.147
30 %	0.218	0.182	0.16	0.153	0.136	0.134	0.134	0.245	0.238	0.235	0.243	0.247	0.248	0.245
40 %	0.317	0.281	0.259	0.251	0.234	0.231	0.232	0.357	0.348	0.343	0.349	0.352	0.352	0.348
50 %	0.418	0.382	0.36	0.352	0.335	0.332	0.333	0.461	0.450	0.445	0.451	0.454	0.454	0.450

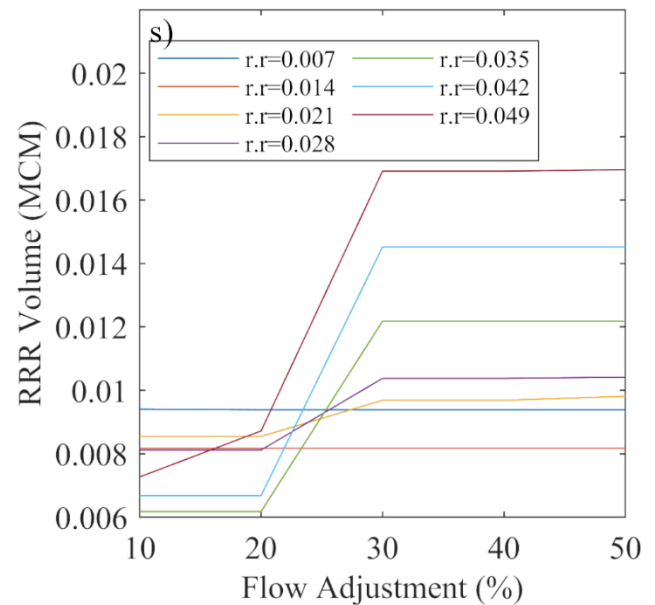
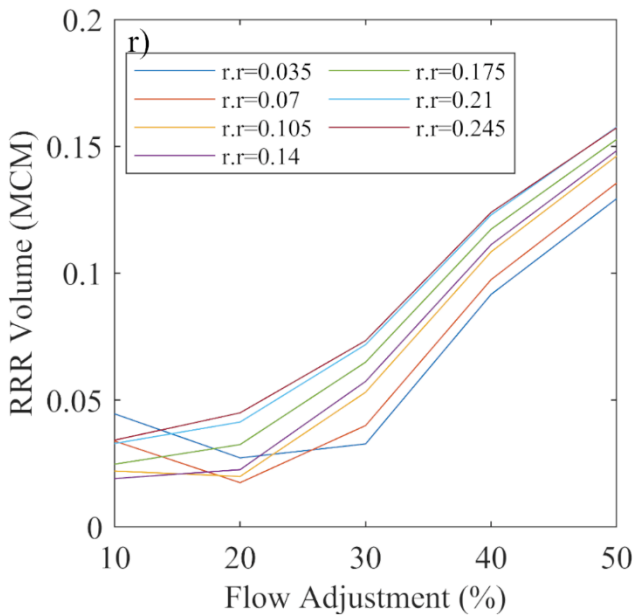
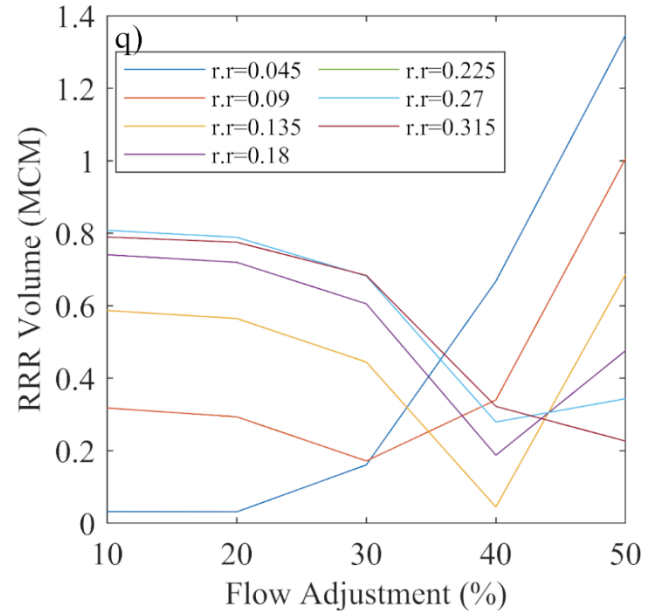
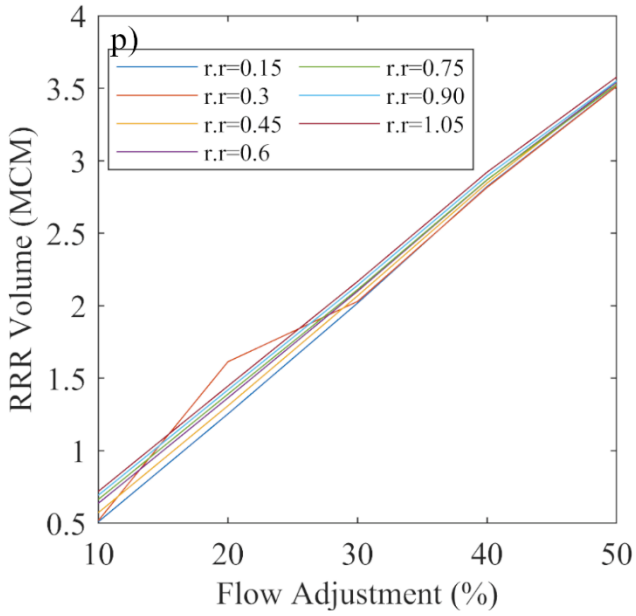
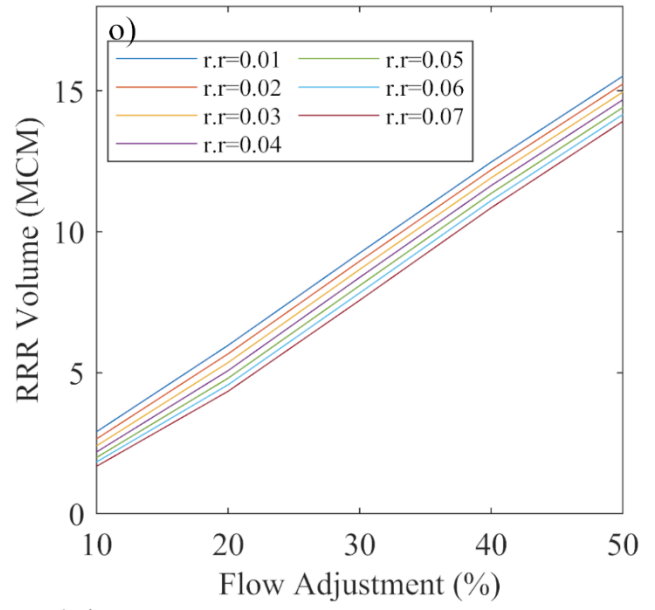
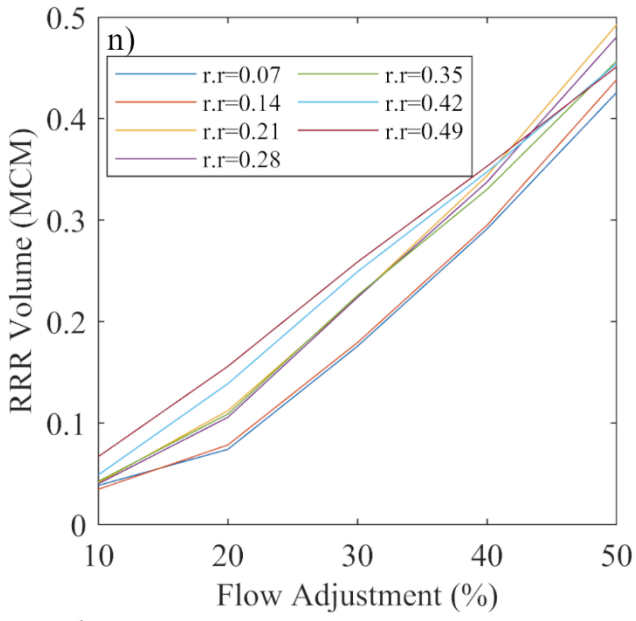
	g) Peltokoski							h) Juvankoski						
	0.070	0.140	0.210	0.280	0.350	0.420	0.490	0.035	0.07	0.105	0.14	0.175	0.21	0.245
10 %	0.238	0.664	0.845	0.825	0.615	0.352	0.197	2.523	1.534	1.15	1.084	1.024	1.008	1.041
20 %	0.276	0.626	0.813	0.812	0.617	0.370	0.226	5.753	4.69	3.987	3.816	3.719	3.64	3.627
30 %	0.736	0.220	0.477	0.594	0.498	0.354	0.292	9.351	8.255	7.524	7.32	7.208	7.107	7.066
40 %	1.581	0.695	0.230	0.079	0.118	0.164	0.175	12.74	11.65	10.96	10.76	10.65	10.54	10.49
50 %	2.394	1.456	0.941	0.658	0.598	0.632	0.611	16.2	15.1	14.43	14.24	14.11	14	13.94
	i) Juntola							j) Kyröskoski						
	0.05	0.1	0.15	0.2	0.25	0.3	0.35	0.700	0.140	0.210	0.280	0.350	0.420	0.490
10 %	2.013	1.212	0.817	0.777	0.763	0.816	0.884	5.613	4.819	4.162	3.751	3.502	3.427	3.404
20 %	4.35	3.432	2.922	2.797	2.689	2.665	2.675	10.644	9.835	9.162	8.738	8.475	8.398	8.372
30 %	7.191	6.238	5.699	5.54	5.413	5.355	5.334	14.987	14.354	13.782	13.366	13.076	12.994	12.975
40 %	10.34	9.346	8.762	8.551	8.381	8.303	8.259	19.458	18.867	18.337	17.957	17.692	17.625	17.625
50 %	13.58	12.49	11.89	11.66	11.47	11.37	11.31	23.716	23.132	22.609	22.239	21.983	21.922	21.931
	k) Sallilankoski							m) Ryötönkoski						
	0.085	0.17	0.255	0.34	0.425	0.51	0.595	0.018	0.035	0.053	0.070	0.088	0.105	0.123
10 %	1.971	2.454	2.708	2.71	2.645	2.591	2.572	0.297	0.273	0.251	0.234	0.221	0.209	0.192
20 %	3.933	3.82	3.813	3.708	3.564	3.423	3.331	0.681	0.657	0.634	0.614	0.596	0.579	0.556
30 %	6.66	6.321	5.967	5.819	5.62	5.436	5.306	1.056	1.030	1.004	0.981	0.961	0.942	0.917
40 %	9.702	9.298	8.782	8.386	8.124	7.925	7.792	1.450	1.422	1.395	1.370	1.347	1.327	1.302
50 %	12.66	12.21	11.65	11.25	10.96	10.73	10.58	1.847	1.815	1.781	1.752	1.726	1.704	1.678
	n) Vatajankoski							o) Niiles						

	0.070	0.140	0.210	0.280	0.350	0.420	0.490	0.01	0.02	0.03	0.04	0.05	0.06	0.07
10 %	0.039	0.035	0.041	0.040	0.043	0.049	0.067	2.904	2.65	2.402	2.186	1.99	1.832	1.679
20 %	0.074	0.079	0.112	0.106	0.109	0.139	0.156	5.971	5.669	5.36	5.073	4.805	4.571	4.341
30 %	0.176	0.179	0.224	0.224	0.226	0.249	0.259	9.246	8.957	8.664	8.38	8.091	7.829	7.569
40 %	0.291	0.295	0.344	0.338	0.330	0.348	0.353	12.47	12.2	11.92	11.64	11.36	11.11	10.85
50 %	0.426	0.438	0.493	0.481	0.457	0.454	0.451	15.52	15.24	14.96	14.68	14.41	14.16	13.92
	p) Kalajärven							q) Hanhikoski						
	0.150	0.300	0.450	0.600	0.750	0.900	1.050	0.045	0.090	0.135	0.180	0.225	0.270	0.315
10 %	0.510	0.518	0.572	0.637	0.663	0.693	0.719	0.032	0.318	0.587	0.740	0.808	0.808	0.789
20 %	1.253	1.613	1.309	1.364	1.390	1.419	1.445	0.031	0.293	0.565	0.720	0.789	0.789	0.775
30 %	2.018	2.029	2.067	2.099	2.111	2.140	2.166	0.161	0.172	0.444	0.605	0.682	0.682	0.683
40 %	2.822	2.816	2.845	2.867	2.868	2.895	2.921	0.667	0.340	0.045	0.187	0.279	0.279	0.322
50 %	3.515	3.512	3.540	3.541	3.525	3.551	3.577	1.346	1.005	0.686	0.476	0.343	0.343	0.226
	r) Vääräkoski							s) Kattilakoski						
	0.015	0.030	0.045	0.060	0.075	0.090	0.105	0.007	0.014	0.021	0.028	0.035	0.042	0.049
10 %	0.045	0.034	0.022	0.019	0.025	0.033	0.034	0.009	0.008	0.009	0.008	0.006	0.007	0.007
20 %	0.027	0.017	0.020	0.023	0.032	0.041	0.045	0.009	0.008	0.009	0.008	0.006	0.007	0.009
30 %	0.033	0.040	0.053	0.057	0.065	0.072	0.073	0.009	0.008	0.010	0.010	0.012	0.015	0.017
40 %	0.092	0.097	0.108	0.111	0.117	0.123	0.124	0.009	0.008	0.010	0.010	0.012	0.015	0.017
50 %	0.129	0.136	0.146	0.148	0.153	0.158	0.157	0.009	0.008	0.010	0.010	0.012	0.015	0.017

	t) Kaitfors							u) Porttipahta						
	0.120	0.240	0.360	0.480	0.600	0.720	0.840	0.165	0.330	0.495	0.660	0.825	0.990	1.155
10 %	0.710	0.654	0.513	0.407	0.249	0.167	0.468	1.460	1.445	1.275	1.071	1.460	0.794	0.684
20 %	1.671	1.426	1.167	1.113	1.073	0.746	0.489	2.909	2.846	2.616	2.355	2.909	2.002	1.864
30 %	2.860	2.606	2.255	2.284	2.170	1.778	1.400	4.893	4.762	4.487	4.167	4.893	3.735	3.585
40 %	4.108	3.789	3.420	3.517	3.400	2.966	2.582	7.041	6.865	6.548	6.200	7.041	5.710	5.553
50 %	5.053	4.751	4.391	4.488	4.373	3.943	3.554	9.036	8.811	8.454	8.080	9.036	7.563	7.395
	v) Kokkosniva							w) Seitakorva						
	0.235	0.47	0.705	0.94	1.175	1.41	1.645	0.920	1.840	2.760	3.700	4.640	5.560	6.480
10 %	0.296	0.232	0.172	0.11	0.13	0.15	0.12	0.415	0.465	0.249	0.223	0.543	0.745	0.764
20 %	0.713	0.624	0.493	0.394	0.305	0.241	0.268	1.419	1.277	0.831	0.284	0.294	0.653	0.790
30 %	1.539	1.416	1.244	1.11	0.982	0.877	0.861	3.472	3.164	2.472	1.680	1.017	0.524	0.289
40 %	2.994	2.8	2.548	2.339	2.139	1.969	1.903	6.087	5.630	4.772	3.807	3.008	2.409	2.087
50 %	4.561	4.283	3.94	3.635	3.356	3.117	3	8.130	7.558	6.562	5.473	4.575	3.894	3.511
	x) Taivalkoski							y) Myllykoski						
	1.00	1.50	2.00	2.50	3.00	3.50	4.00	0.025	0.05	0.075	0.1	0.125	0.15	0.175
10 %	0.453	0.362	0.260	0.147	0.143	0.256	0.368	0.052	0.073	0.087	0.119	0.155	0.187	0.21
20 %	0.818	0.757	0.675	0.573	0.444	0.303	0.262	0.102	0.118	0.13	0.161	0.197	0.228	0.251
30 %	1.558	1.514	1.448	1.359	1.241	1.116	0.982	0.161	0.173	0.184	0.215	0.251	0.282	0.305
40 %	2.583	2.555	2.504	2.434	2.338	2.232	2.120	0.196	0.205	0.215	0.246	0.282	0.313	0.336
50 %	3.572	3.556	3.526	3.483	3.418	3.344	3.262	0.212	0.222	0.232	0.263	0.299	0.33	0.353







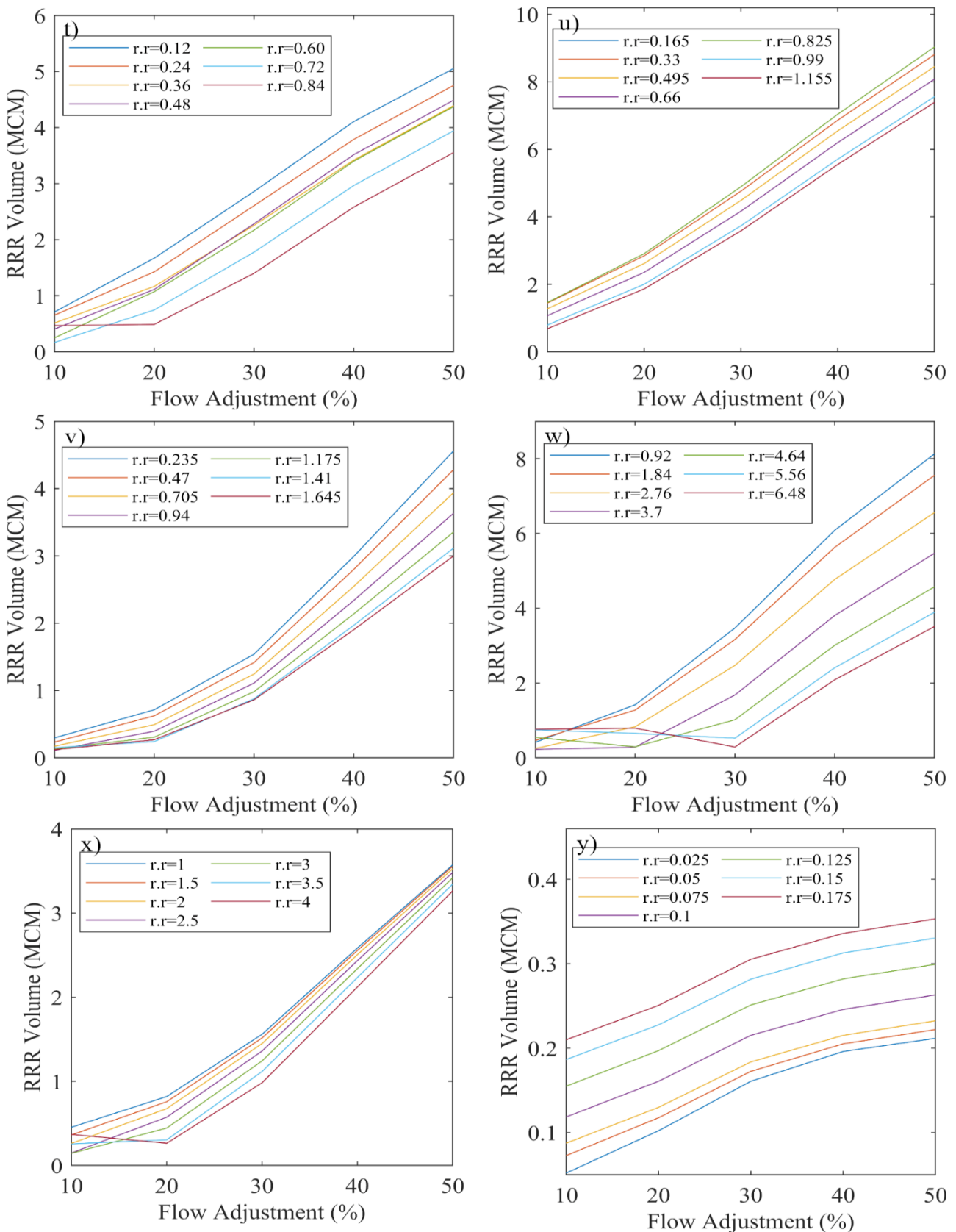


Figure 3. Required RRR volume for different flow and ramp rate thresholds (r.r is ramp rate). a) Lieksankoski b) Pamilon c) Leuhunkoski d) Hietamankoski e) Kannuskoski f) Anjalankosken g) Peltokoski h) Juvankoski i) Juntola j) Kyröskoski k) Sallilankoski m) Ryötönkoski n) Vatajankoski o) Niiles p) Kalajärven q) Hanhikoski r) Vääräkoski s) Kattilakoski t) Kaitfors u) Porttipahta v) Kokkosniva w) Seitakorva x) Taivalkoski y) Myllykoski.

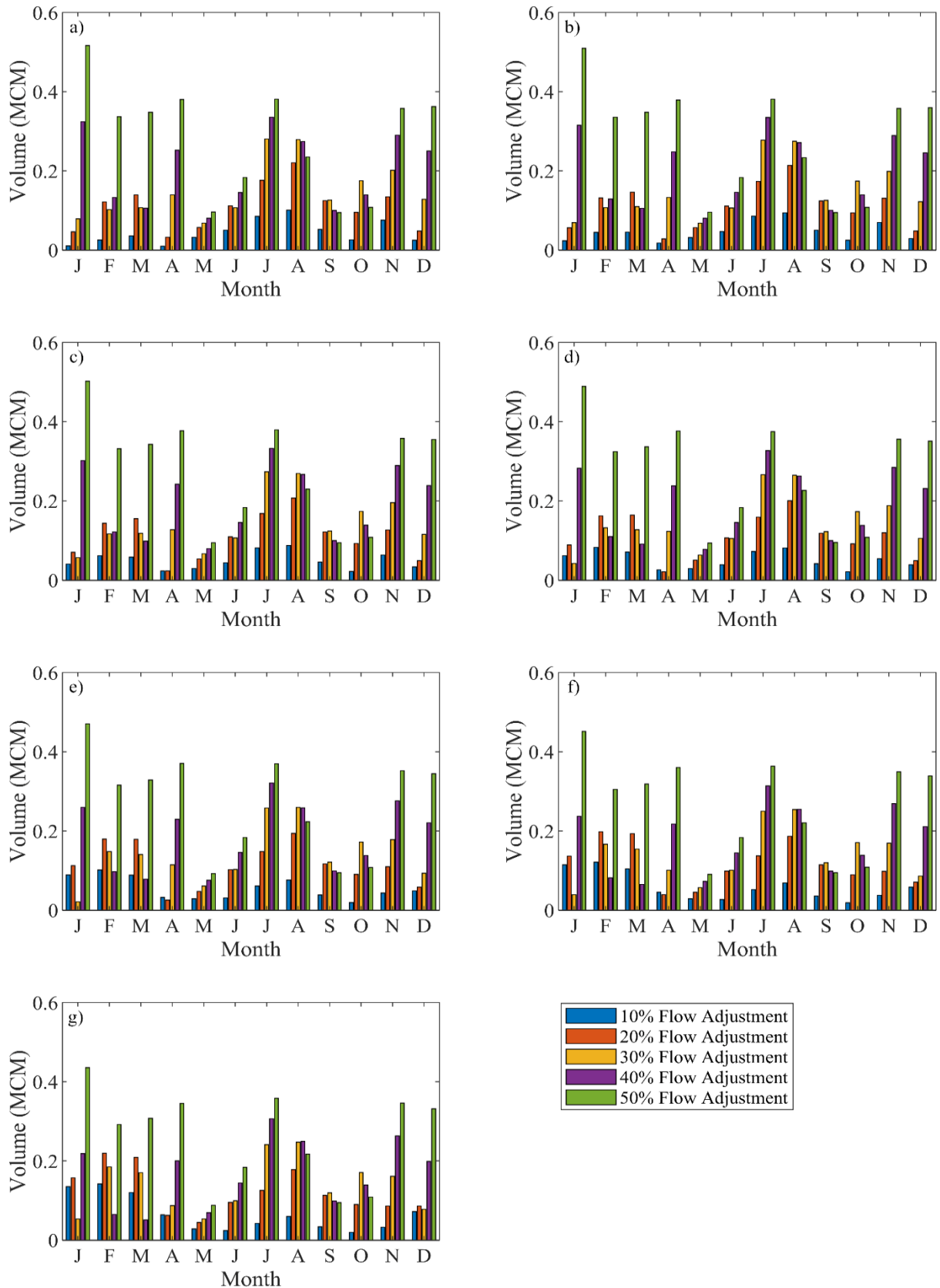


Figure 4. The required monthly volume of RRR for Taivalkoski (scaled flow), without carryover of stored water from one month to the other, if there is any. The r.r for a) 1, b) 1.5, c) 2, d) 2.5, e) 3, f) 3.5, g)  $4 \text{ m}^3\text{s}^{-1}\text{min}^{-1}$

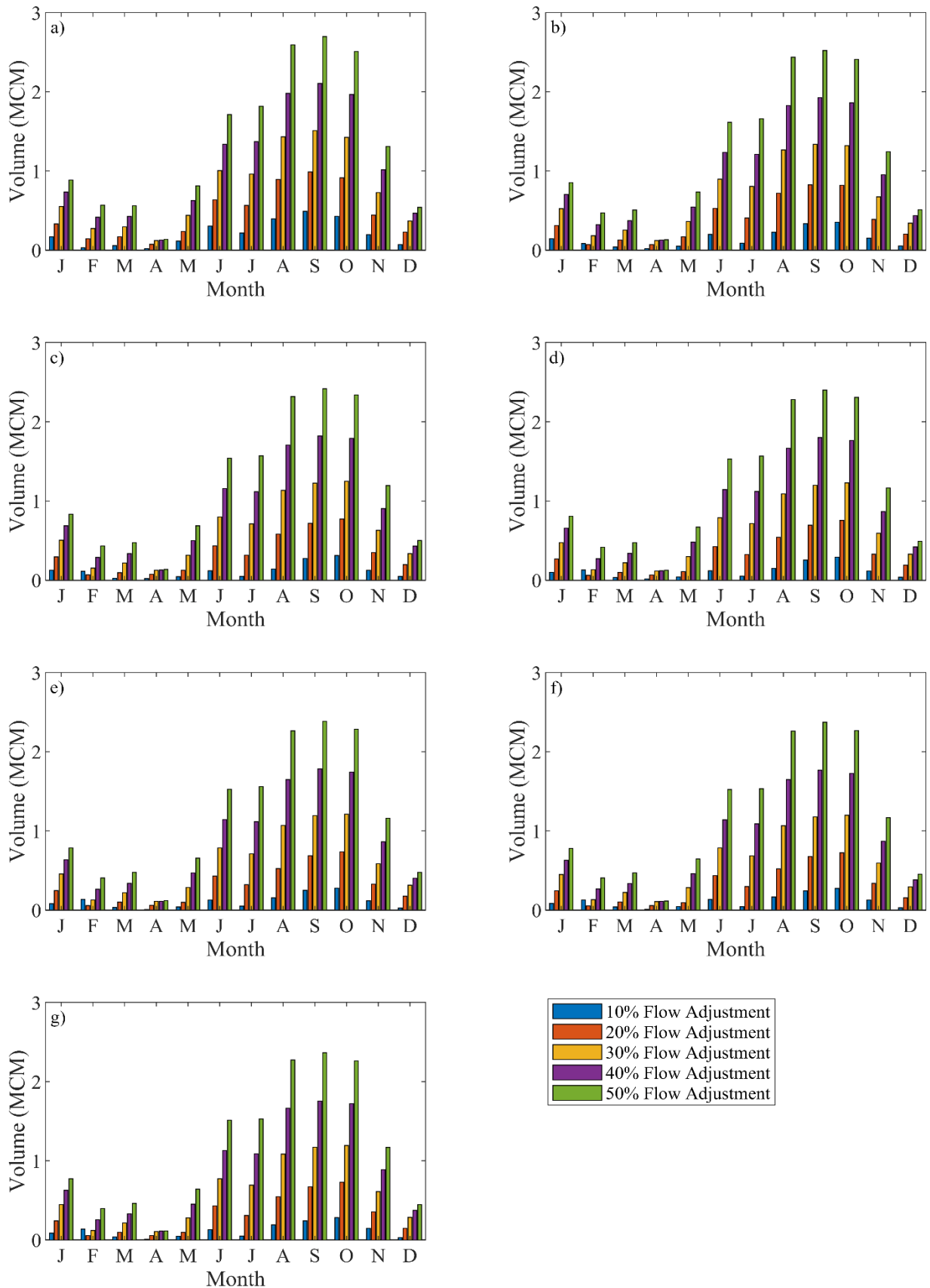


Figure 5. The required monthly volume of RRR for Juvankoski (scaled flow), without carryover of stored water from one month to the other, if there is any. The r.r for a) 0.035, b) 0.070, c) 0.105, d) 0.140, e) 0.175, f) 0.210, g) 0.245  $\text{m}^3\text{s}^{-1}\text{min}^{-1}$

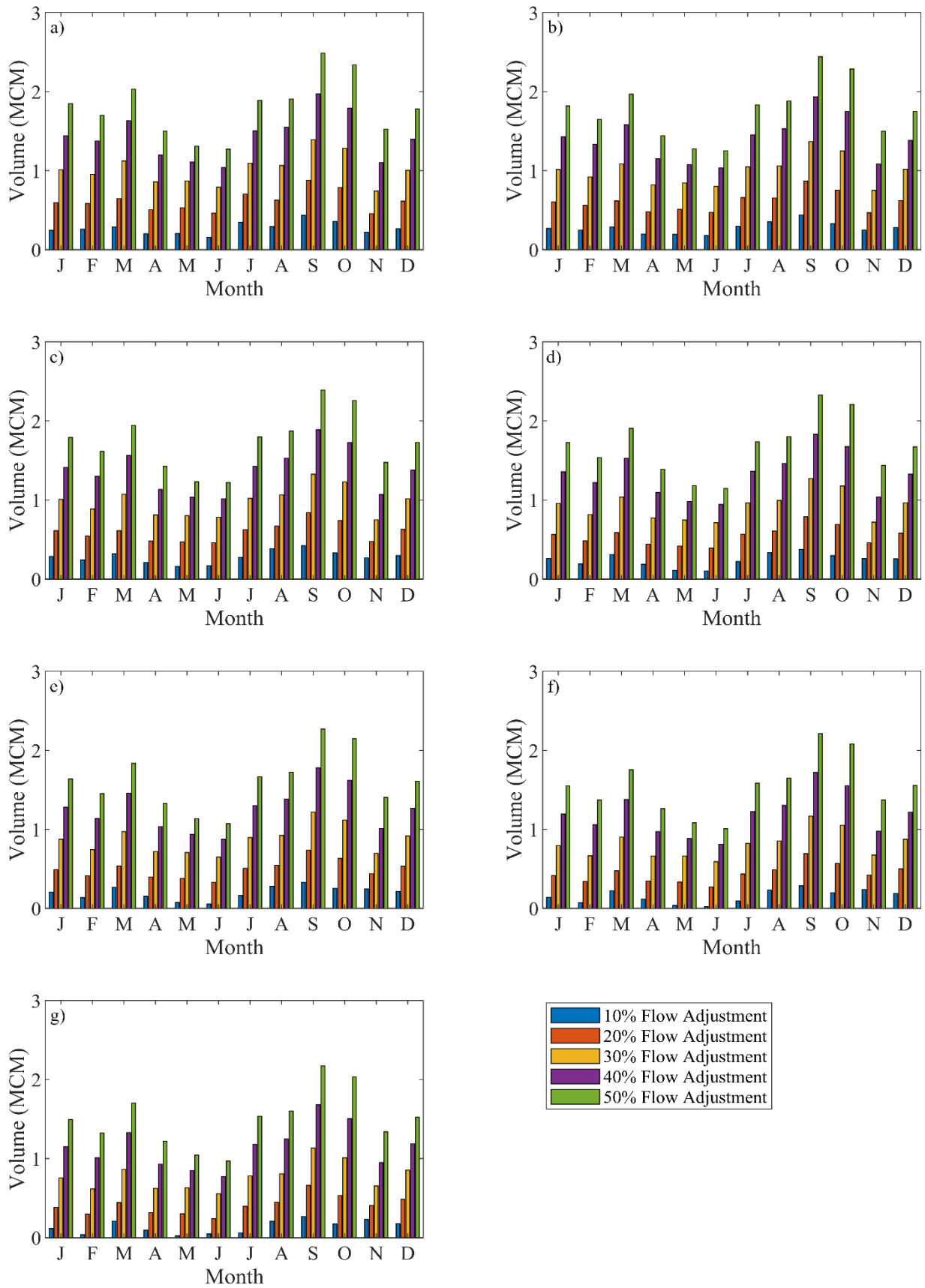


Figure 6. The required monthly volume of RRR for Pamilon (scaled flow), without carryover of stored water from one month to the other, if there is any. The r.r for a) 0.210, b) 0.420, c) 0.630, d) 0.840, e) 1.050, f) 1.260, g)  $1.470 \text{ m}^3\text{s}^{-1}\text{min}^{-1}$

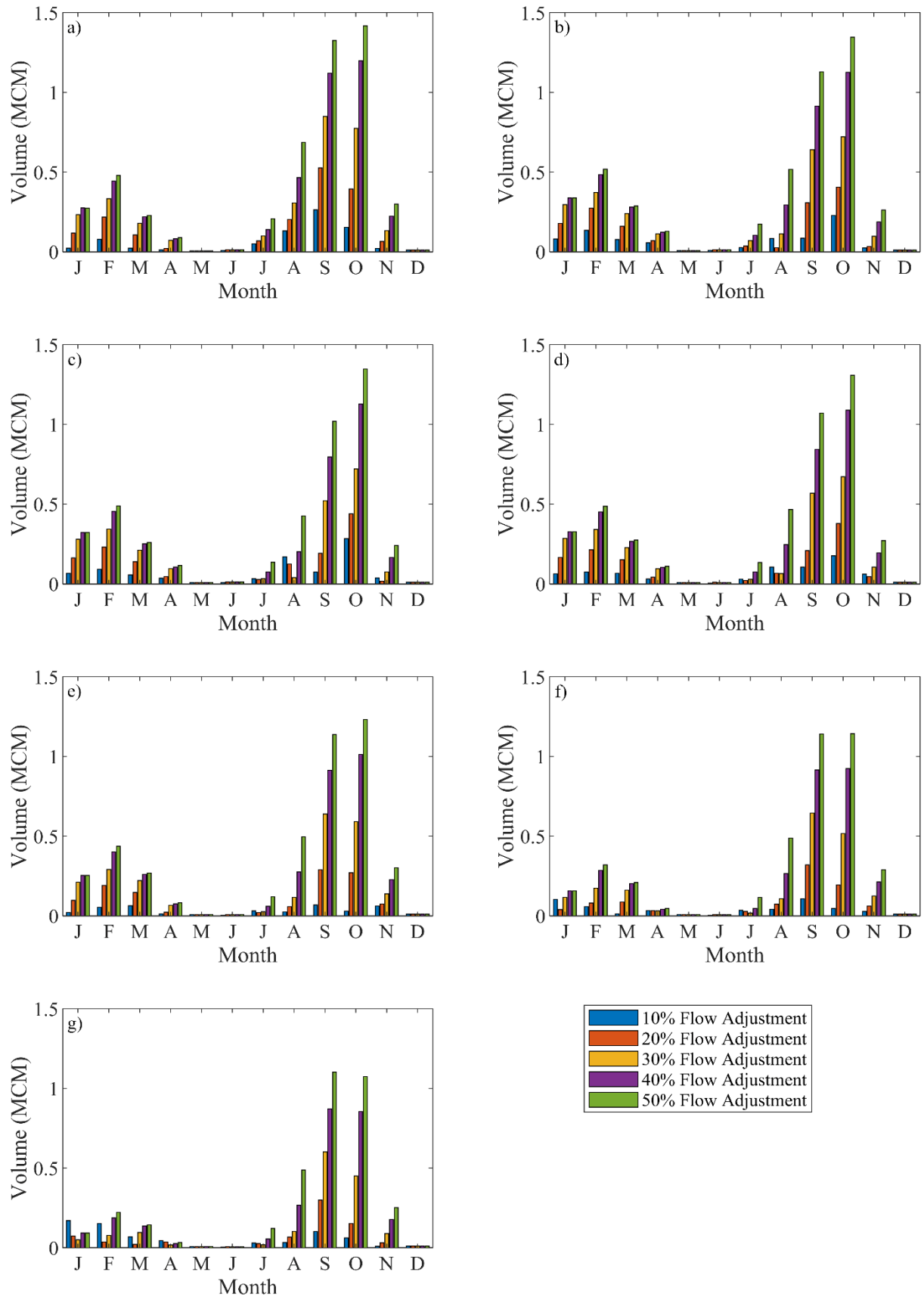


Figure 7. The required monthly volume of RRR for Kaitfors (scaled flow), without carryover of stored water from one month to the other, if there is any. The r.r for a) 0.120, b) 0.240, c) 0.360, d) 0.480, e) 0.60, f) 0.720, g) 0.840  $\text{m}^3\text{s}^{-1}\text{min}^{-1}$

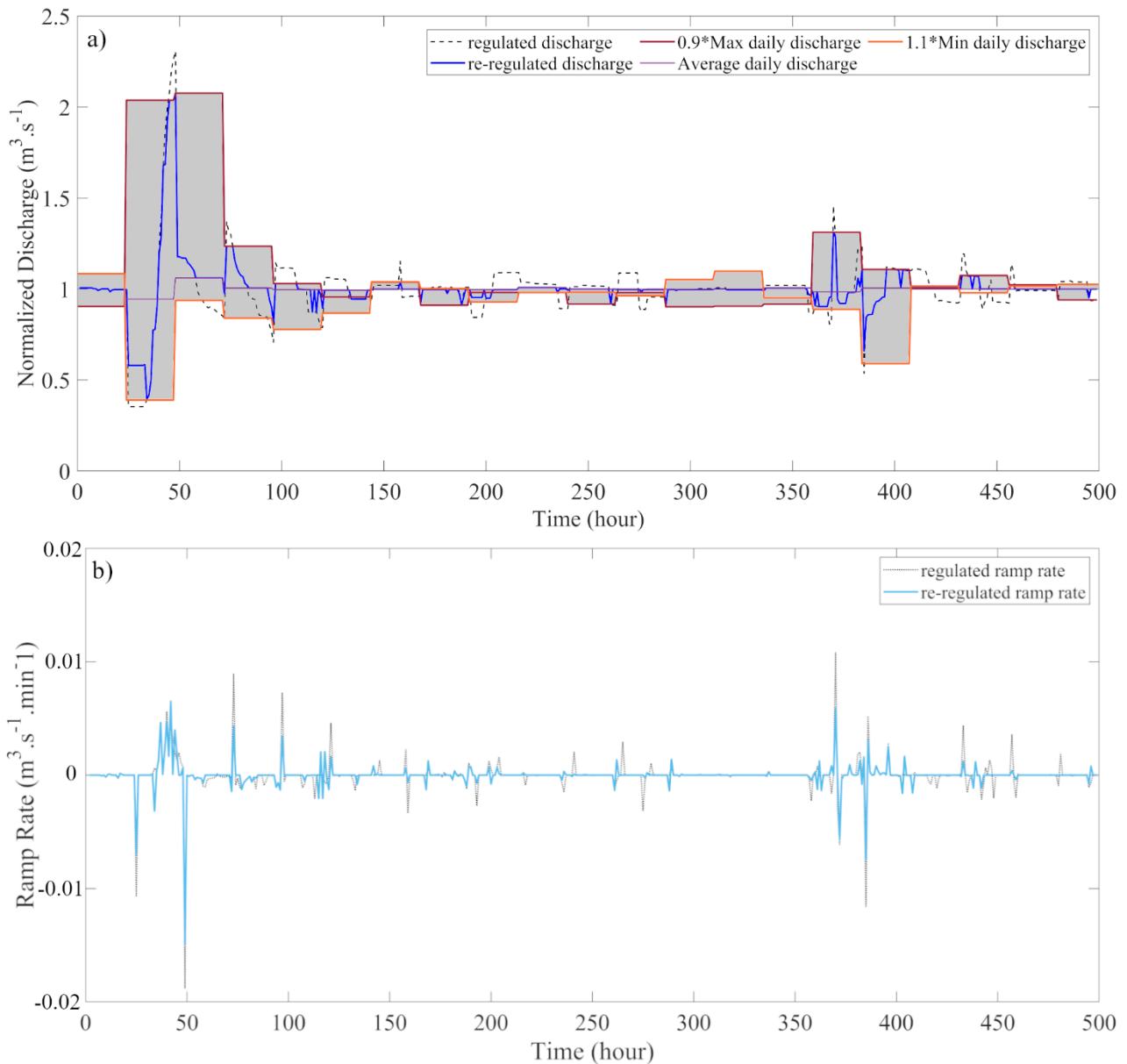


Figure 8. The performance of the re-regulation reservoirs to re-regulate the discharge for Juvankoski P-h (10%, 0.035) for 500 hours. (a) illustrates how the reservoir limits the discharge to the thresholds defined in the algorithm and achieves optimal conditions whenever possible. (b) distinctly shows the ramping rates before and after implementing the re-regulation reservoir.

## 6. DISCUSSION

RRRs have emerged as a potentially effective approach for mitigating the adverse ecological impacts of hydropeaking. Although the efficacy of RRRs in mitigating hydropeaking impacts is still under investigation, initial studies indicate that this novel solution may be a promising approach to enhance the ecological sustainability of hydropower operations. Empirical research has demonstrated that implementing RRRs can mitigate the adverse downstream effects of hydropeaking caused by HPPs and reduce their detrimental impact on aquatic ecosystems (Tonolla et al., 2017).

Compared to alternative mitigation measures, such as downstream flow control devices or operational modifications to hydropower facilities, RRRs present several advantages. These include enhanced flexibility and adaptability in response to non-stationary environmental conditions, and varying power and water demand, without a corresponding increase in power system operational costs. Furthermore, RRRs can provide additional benefits, such as water storage and flood control (Anindito et al., 2019; Premstaller et al., 2017; Tonolla et al., 2017). In practical applications, RRRs could be implemented using several water storage options such as artificial structures, natural water storage, floodplains, or other nearby storage alternatives. Our study did not give precedence to evaluating the structural configurations of RRRs. Instead, our focus was to investigate theoretical methodologies for mitigating hydropeaking and to develop a model that can efficiently operate and estimate the necessary volume of RRRs. This study developed a re-regulation algorithm to determine the volumes of RRRs required to achieve the stated objectives and priorities in section 6. The analysis was conducted on 24 stations located in Finnish rivers. To analyze the impact of various flow patterns on the required RRR volumes, a base case was generated by scaling down the hourly flow data of 24 stations. Although re-regulation practices are case specific due to the distinct flow patterns of each river, our case data from 24 stations of various river sizes can still be leveraged to explore theoretical possibilities for re-regulation practices.

The findings of this study demonstrate that in most of the tested stations and permutations, more stringent peak and minimum hourly flow thresholds, as well as ramping rates led to a corresponding increase in the required RRRs volume (Figure 3). Most stations had a positive linear correlation between the required volume of the RRRs and the thresholds (see Table 2). The observed outcome can be attributed to two factors. Firstly, as the

thresholds became more stringent, they activated the conditions that enable water storage by the algorithm at more time steps than those that facilitate water release. The flow pattern at any given station controls the algorithm's decision to store or release water. Secondly, a more stringent threshold necessitates storing or releasing larger quantities of water. Considering both factors, i.e., increased water storage events and an increased water volume to be stored in a singular event, led to the requirement of greater RRR volumes as the thresholds became more stringent at these stations. Nonetheless, this trend was not observed for several permutations at some stations. Several station permutations did not show a straightforward relationship between thresholds and required RRR volumes (See Table 2). For stations (a), (e), (r), and (t), several permutations with 10% flow adjustment required a smaller RRR volume than permutations with 20% flow adjustment. For station (g), when the flow adjustment further increased from 20% to 30%, the required RRR volume decreased for all permutations, to then increase when the flow adjustment was set above 30%. Contrary to stations that displayed a positive linear relationship, the increased flow adjustment threshold in these stations activated the conditions that enable water release by the algorithm at more time steps than those that allow water storage. Thus, resulting in a reduction of the required RRR volumes. This highlights the significance of maintaining a proportional balance between storing and releasing water. However, this may not always be feasible due to the conflict between the restoration objectives and the unique flow patterns of the river.

Apart from the flow adjustment threshold, the correlation between the ramping rate threshold and the required RRR volume also displayed non-linear relationships. One example where this is evident is at station (k), where P-k (20%, 0.510) required a larger RRR volume than permutations with a more stringent r.r threshold. In such cases, a greater RRR volume was required due to the increased propensity of the algorithm to store water rather than release water. Thus, it is possible for the water released in proportion to the water stored for permutations with less stringent thresholds to be less efficient than those with stricter thresholds. Therefore, inadequate (i.e., too slow) water release back into the waterway increases the required RRR volume or leads to small volume availability during high flows. The relationship between the required RRR volume, daily peak discharge, and ramp rate thresholds for the 24 selected stations is demonstrated in Figure 3. Furthermore, stations exhibiting non-linear relationships between thresholds and RRR volume requirement highlights the importance of matching

the flow adjustment and ramping rate thresholds. Adequate matching of the thresholds allows for optimal flow pattern restoration. One example is at station (x); if a 10% flow adjustment is the targeted objective, then it might be more ecologically beneficial to consider P-x (10%, 2.5) than P-x (10%, 3) since it has more stringent ramp rate thresholds while requiring a smaller RRR volume. Another example is at station (a), where a greater flow adjustment could be achieved without compromising the ramping rate thresholds. P-a (20%, 7) requires a smaller RRR volume than P-a (10%, 7) while having a greater flow adjustment and the same ramping rate thresholds. Therefore, selecting the ideal volume for RRRs is closely linked to the flow pattern of the river and its sub-daily flow dynamics. These observations underscore the criticality of considering a particular river's distinctive features and its surrounding ecosystem when designing RRRs to mitigate the impacts of hydropeaking.

Furthermore, each month's required RRR volume was determined separately at Taivalkoski, Juvankoski, Pamilon, and Kaitfors stations, illustrated in Figures 4, 5, 6, and 7. For most of the investigated permutations at Taivalkoski station, except permutations with 50% flow adjustment, July and August require the largest RRR volume to achieve the objectives and priorities in section 4. Whereas, for permutations with a 50% flow adjustment, January is the month that requires the largest reservoir volume. However, it is essential to note that January and February are the months with the largest volume requirement when the ramp rate thresholds exceed  $2.5 \text{ m}^3 \cdot \text{s}^{-1} \cdot \text{min}^{-1}$  for permutations with a 10% flow adjustment. On the other hand, for stations Juvankoski, Pamilon, and Kaitfors, September and October require the largest RRR volume for all the permutations. The large RRR volume requirement for these months is due to the highest flow occurring in late spring or early summer.

To enhance the efficiency of RRRs, customization of their design according to the ecological requirements of the river and the mitigation objectives can be employed. Hayes et al. (2019) proposed a species-specific mitigation approach that targets the life cycle stages of fish. The ecological significance of seasons, days, or time steps can be considered when setting thresholds and objectives, leading to reduced RRRs volume requirements by excluding periods with less ecological importance from the operation period. In the context of the present study, the most substantial reduction in required RRR volume for Taivalkoski station can be achieved by excluding July and August from the operation period, and September and October from the operation period of Juvankoski,

Pamilon, and Kaitfors stations. Therefore, optimizing the RRRs operation period can lead to greater efficiency. However, to minimize the required RRR volume, the re-regulation operation period should be customized according to the ecological necessities of the river. If the goal is to enhance the larval stage of trout and grayling instead of eradicating fish stranding, a less stringent ramp rate threshold could be more productive, resulting in decreased RRRs volume requirements. In general, these findings demonstrate the potential of optimizing the design and operation of RRRs for hydropeaking mitigation based on the distinct needs of the local ecosystem and the desired mitigation objectives. By doing so, it is possible to achieve greater efficiency, reduce the required RRR volume, and minimize the downstream ecosystem's impact of hydropeaking.

Additionally, the RRR operation for P-h (10%, 0.035) for the scaled flow downstream of Juvankoski is demonstrated in Figure 8. For all the permutations and stations, the RRR limits peak and minimum hourly flows and ramp rates according to thresholds defined by the algorithm. Nevertheless, Figure 8.a. demonstrates that the RRR increases the minimum flow beyond the specified threshold in numerous time steps without violating other priorities. However, as illustrated in Figure 8.b., there are time steps where priority 2 takes precedence over priority 3, causing the ramp rates to surpass the defined threshold.

The model developed in this study has the potential to determine the necessary RRR volumes for other HPPs in river systems of varying sizes and flows. However, the thresholds for hydropeaking must be appropriately defined to match the flow patterns and river dimensions. Additionally, the model developed in this study does not account for the time needed for the water to flow from the HPP to the RRR and for the water losses that might occur, such as evaporation and seepage. Thus, incorporating flow velocity and water losses from the RRR would enhance the model's accuracy. Furthermore, integrating additional conditions into the model, such as RRR operational period (season), hydropeaking timing (day and night), water temperature fluctuations, and sediment loads, can expand the RRRs scope. The model could be further enhanced by accounting for water supply, irrigation, and recreational needs. Furthermore, investigating a model with a RRR located far from the HPP would allow for examining corridors that connect HPPs with RRRs as supplementary storage volumes. These enhancements would allow for better comprehension, usage, and optimization of RRRs.

## **7. CONCLUSION**

The reviewed literature in this study highlights the diverse ecological impacts of flow alteration and hydropeaking on ecosystems, ranging from changes in habitat availability to alterations in species composition and biodiversity. Flow alteration leads to the degradation of water quality and the decline of aquatic populations, with severe consequences for ecosystem function and services. These findings emphasize the importance of mitigating the adverse impacts of flow alteration and hydropeaking. Further research is needed to better understand the complex interactions between flow alteration and ecological systems, particularly when considering environmental non-stationarity. Overall, this literature review highlights the urgent need for a mitigation measure that considers the ecological consequences of hydropeaking without adding operating costs to HPPs.

In this study, we examined the potential of RRRs to restrict flow components to specific thresholds, including peak flow, minimum flow, and ramping rates. For most stations, the results indicated that the required RRR volume had a positive linear correlation with the flow adjustment and ramping rate thresholds. Thus, as the thresholds become more stringent, the required RRR volume increases. However, this trend was not observed for some stations, indicating that inadequate (i.e., too slow) water release into the waterway might increase the required RRR volume or lead to small availability during high flows. Moreover, the results show that achieving greater flow pattern restoration is possible without increasing the RRR volume by adequately matching the flow adjustment and ramping rate thresholds. However, the design of RRRs should be tailored according to hydropeaking mitigation objectives and the ecological needs of river systems. The model developed in this study can assist HPP operators, authorities, and researchers in the design, optimization, and feasibility assessment of introducing RRRs to river systems. Nonetheless, several potential research pathways warrant exploration to enhance comprehension, application, and optimization of RRRs. Among these, a particularly promising avenue is the advancement of more sophisticated models that integrate variables such as RRR operational period (season), timing of hydropeaking events (day or night), as well as fluctuations in water temperature and sediment loads. In conclusion, further research is required to improve our comprehension of the ecological consequences of hydropeaking, encompassing its impacts on fish populations, water quality, and overall ecosystem health. By enhancing our understanding of hydropeaking's impact, potential mitigation measures, and RRRs, we can devise and optimize more efficient approaches

for managing water resources and reducing the adverse impacts associated with hydropeaking.

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