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1 **Review: Impacts of permafrost degradation on inorganic chemistry of surface**
2 **fresh water**

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16
17 **Abstract**

18 Recent studies have shown that climate change is impacting the inorganic chemical characteristics of
19 surface fresh water in permafrost areas and affecting aquatic ecosystems. Concentrations of major ions
20 (e.g., Ca²⁺, Mg²⁺, SO₄²⁻, NO₃⁻) can increase following permafrost degradation with associated deepening
21 of flow pathways and increased contributions of deep groundwater. In addition, thickening of the active
22 layer and melting of near-surface ground ice can influence inorganic chemical fluxes from permafrost into
23 surface water. Permafrost degradation has also the capability to modify trace element (e.g., Ni, Mn, Al,
24 Hg, Pb) contents in surface water. Although several local and regional modifications of inorganic
25 chemistry of surface fresh water have been attributed to permafrost degradation, a comprehensive review
26 of the observed changes is lacking. The goal of this paper is to distil insight gained across differing
27 permafrost settings through the identification of common patterns in previous studies, at global scale. In

28 this review we focus on three typical permafrost configurations (pervasive permafrost degradation,
29 thermokarst, and thawing rock glaciers) as examples and distinguish impacts on (i) major ions and (ii)
30 trace elements. Consequences of warming climate have caused spatially-distributed progressive
31 increases of major ion and trace element delivery to surface fresh water in both polar and mountain areas
32 following pervasive permafrost degradation. Moreover, localised releases of major ions and trace
33 elements to surface water due to the liberation of soluble materials sequestered in permafrost and ground
34 ice have been found in ice-rich terrains both at high latitude (thermokarst features) and high elevation
35 (rock glaciers). Further release of solutes and related transport to surface fresh water can be expected
36 under warming climatic conditions. However, complex interactions among several factors able to influence
37 the timing and magnitude of the impacts of permafrost degradation on inorganic chemistry of surface fresh
38 water (e.g., permafrost sensitivity to thawing, modes of permafrost degradation, characteristics of
39 watersheds) require further conceptual and mechanistic understanding together with quantitative
40 diagnosis of the involved mechanisms in order to predict future changes with confidence.

41

42 **1. Introduction**

43 Average atmospheric temperature has increased globally over the last decades and, as a response,
44 the cryosphere is changing (Fountain et al., 2012). Permafrost, a component of the cryosphere, is
45 widespread in the Arctic and Antarctica, and in cold mountains, including densely populated areas of the
46 European Alps and Asian mountain ranges (Gruber, 2012). Permafrost interacts with climate (Walter
47 Anthony et al., 2006; Schuur et al., 2015), hydrology (e.g., Liljedahl et al., 2016; Yang et al., 2017),
48 ecosystems (Jorgenson et al., 2001; Vonk et al., 2015), geophysical processes (e.g., Gruber et al., 2004;
49 Gruber and Haeberli, 2009), and human systems (Nelson et al., 2002; Harris et al., 2009).

50 Recent reviews have focused on the impacts of permafrost warming and degradation on river
51 biogeochemistry (Frey and McClelland, 2009) and aquatic ecosystems (Vonk et al., 2015) in the Arctic.
52 Notwithstanding, local and regional modifications of water hydrochemistry due to permafrost degradation
53 have been reported from many locations, globally. Given the sparsity of data available, understanding and
54 analysing permafrost degradation impacts on inorganic chemistry of surface fresh water will benefit from

55 identifying common patterns in existing studies. The present review thus aims to distil insight gained
56 across differing permafrost environments and configurations globally.

57 Following a brief background section, we distinguish three typical example configurations of permafrost
58 thaw. For those, we review impacts of permafrost degradation on major ions (e.g., Ca^{2+} , Mg^{2+} , SO_4^{2-} , NO_3^-
59) and on trace elements (e.g., Ni, Mn, Al, Hg, Pb).

60

61 **2. Permafrost: definition and main characteristics**

62 Permafrost is defined as ground (soil and/or rock, including ice and organic material, plus air/gas in
63 unsaturated ground) that remains at or below 0 °C for at least two consecutive years (Muller, 1943). Thus,
64 permafrost is a thermal phenomenon and it can, but does not need to, contain water or ice. Most
65 permafrost areas experience seasonal thaw, during which ground surface temperature rises above 0 °C.
66 The layer of ground that thaws on a seasonal basis is called “active layer” (ACGR, 1988). The depth of
67 permafrost (permafrost base) is determined by the geothermal heat flux, subsurface properties, and the
68 long-term surface temperature. The upper limit of permafrost (permafrost table) is largely controlled by
69 surface temperatures during summer and near-surface ground properties (Williams and Smith, 1989;
70 French, 2007). The difference in depth between the permafrost table and the permafrost base is the
71 permafrost thickness (see Figure 2 in ACGR, 1988, source: <https://ipa.arcticportal.org>). Permafrost
72 regions are often divided into a continuous permafrost zone, where more than ~90% of the surface area is
73 underlain by permafrost, a discontinuous zone, of which ~10% to ~90% is underlain by permafrost, and a
74 zone of isolated patches of permafrost, where permafrost occurs under less than ~10% of the ground. The
75 discontinuous zone is often divided into extensive discontinuous permafrost (~90% to ~50%) and sporadic
76 discontinuous permafrost (~50% to ~10%, Heginbottom et al., 1995).

77 The global permafrost region (exposed land surface below which some permafrost can be expected) is
78 estimated to be $22 \pm 3 \times 10^6$ km², approximately 17% of the global land area (Gruber, 2012). Zhang et al.
79 (1999; 2003) estimated that permafrost could underlie nearly 25% of the exposed land area of the
80 Northern Hemisphere. Despite occupying only 0.36% (49,800 km²) of the Antarctic region, permafrost is
81 estimated to be present beneath all ice-free terrain, except at the lowest elevations of the maritime
82 Antarctic and sub-Antarctic islands (Vieira et al., 2010). Although with increased spatial heterogeneity

83 imposed by topography, permafrost is also widespread in many mountain ranges such as the European
84 Alps (e.g., Boeckli et al., 2012), the Hindu Kush Himalaya (e.g., Gruber et al., 2017), and the Andes (e.g.,
85 Brenning, 2005).

86 The aggradation and degradation of permafrost usually occur in association with a change of mean
87 ground temperature due to microclimatic (ground cover) and climatic changes (Williams and Smith, 1989).
88 *Permafrost degradation* refers to the decrease of volume of a body of permafrost, whereas *permafrost*
89 *thaw* or *thawing* is used to refer to increasing liquid water content and other changes of physical
90 characteristics in response to energy input within a body of rock or soil while still satisfying the definition of
91 permafrost. The term *thawed permafrost* indicates ground that no longer is permafrost. Global warming
92 over the past several decades is now large enough that regional climate change is emerging above the
93 noise of natural variability (Hansen and Sato, 2016). Despite the small magnitude of warming relative to
94 weather fluctuations, effects of the warming already have notable environmental, social and economic
95 impacts. In this context, the degradation of permafrost is seen as a major challenge in the current
96 discussion of globally rising air temperatures (IPCC, 2013), because of the possible effects on climate
97 (e.g., Harden et al., 2012; Schuur et al., 2015), infrastructure stability (e.g., Nelson et al., 2001; Streletskiy
98 et al., 2012), hydrology (e.g., Liljedahl et al., 2016; Walvoord and Kurylyk, 2016), natural hazards in
99 mountainous areas (e.g., Gruber et al., 2004; Harris et al., 2009), and aquatic biogeochemistry and
100 ecosystems (e.g., Frey and McClelland, 2009; Vonk et al., 2015).

101

102 **3. Typical configurations of permafrost thaw**

103 We focus on three typical configurations in which permafrost degradation can impact the inorganic
104 chemistry of water bodies. While not exhaustive, these examples currently reflect major interests in this
105 research field:

- 106 i) Pervasive permafrost degradation, leading to spatially-distributed modifications in groundwater-
107 surface water connectivity and causing volumes of permafrost to thaw and become subject to
108 weathering and leaching;
- 109 ii) thermokarst and associated localised mobilisations of sediments and solutes;
- 110 iii) rock-glacier thawing and related localised export of solute-rich water.

111 (i) Pervasive permafrost degradation. Permafrost thaw affects hydrology through increasing hydrologic
112 permeability with ice loss in pores (Bense et al., 2009; Kurylyk et al., 2014) and through the release of
113 water stored in frozen material. Permafrost is commonly an aquitard (Woo, 2012; Kane et al., 2013),
114 supporting perched water tables and increasing near-surface soil moisture (Gorbunov, 1978; Ishikawa et
115 al., 2005). Permafrost thaw, through talik growth and higher permeability of partially-frozen material,
116 changes the hydrologic connectivity in a catchment and affects the rates and amounts of flow along
117 differing paths (Hinzman et al., 2005; Rogger et al., 2017), which often produce water with distinct
118 geochemical signatures (Clark et al., 2001).

119 Due to thaw and degradation, new hydrological pathways in permafrost are expected to enhance the
120 interaction between mineralised groundwater and surface water, driving transitional permafrost
121 environments from surface-water dominated systems to groundwater-dominated systems (e.g., Frey and
122 McClelland, 2009). Indeed, recent studies have attributed increases in groundwater contributions to base
123 flow of several rivers in permafrost watersheds to permafrost degradation (Walvoord and Striegl, 2007; St.
124 Jacques and Sauchyn, 2009; Kolosov et al., 2016).

125 Most near-surface effects of permafrost thaw stem from increasing Active Layer Thickness (ALT),
126 which has been shown to occur at large scale, across entire landscapes (e.g., Åkerman and Johansson,
127 2008). Recently, Luo et al. (2016) investigated the spatiotemporal characteristics of ALT across the
128 Northern Hemisphere from 1990 to 2015. Significant trends of increasing ALT have been observed at
129 approximately 43 % of the investigated 169 sites analysed. Regionally, increasing ALT have been found in
130 North America (e.g., Oelke et al., 2004; Burn and Zhang, 2009), Asia (e.g., Frauenfeld et al., 2004; Oelke
131 et al., 2004; Wu et al., 2012), Antarctica (e.g., Guglielmin and Cannone, 2012; Guglielmin et al., 2014),
132 and Europe (e.g., Åkerman and Johansson, 2008; Harris et al., 2009), although variation exists due to
133 local conditions.

134 Increasing ALT can expose newly-thawed soils to near-surface water and increase chemical fluxes
135 from freshly-exposed strata (e.g., Keller et al., 2007; Keller et al., 2010). Furthermore, geochemical
136 constituents previously sequestered within the near-surface permafrost (e.g., Kokelj and Burn, 2005) can
137 be released.

138 Although spatially-distributed modifications in groundwater-surface water connectivity and increasing
139 soil volumes subject to weathering and leaching are distinct processes associated with permafrost
140 degradation, the impacts of these processes on the inorganic chemistry of surface water are not easily
141 discernible. Indeed, both result in diffuse increases of geochemical fluxes into the receiving water bodies
142 and only few studies distinguish the main sources of such fluxes (e.g., Keller et al., 2007, 2010;
143 Lamhonwah et al., 2017).

144 (ii) Thermokarst. Permafrost warming and increasing ALT can cause self-reinforcing thaw of ice-rich
145 ground leading to the development of thermokarst (Jorgenson and Osterkamp, 2005; Rowland et al.,
146 2010). The term thermokarst includes a suite of processes and landforms that involve lowering, and often
147 collapsing, of the land surface because of melting excess ground ice. Kokelj and Jorgenson (2013)
148 classified the principal thermokarst types as: a) hillslope processes, including retrogressive thaw slumps,
149 active layer detachment slides, and thermal erosion gullies; b) thaw lake processes, including lake
150 expansion/reduction, drainage, and lake basin evolution; c) wetland processes, including peatland
151 collapse and the development of bogs and fens. Thermokarst can also occur on active rock glaciers
152 (details in (iii) Rock glacier thawing), where increased melting of permafrost ice can cause the formation of
153 small thermokarst lakes (Kääb and Haeberli, 2001). Thermokarst is commonly observed throughout most
154 permafrost regions including Canada (e.g., Watanabe et al., 2011), Alaska (e.g., Andresen and Lougheed,
155 2015), Russia (e.g., Manasypov et al., 2014), Mongolia (e.g., Sharkhuu, 1998), China (e.g., Wu et al.,
156 2010), Antarctica (e.g., Gooseff et al., 2016) and the European Alps (e.g., Kääb and Haeberli, 2001).

157 Thermokarst development has the potential to alter the geochemical signature of surface water
158 because it facilitates mobilisation of sediments and solutes previously sequestered in ground ice (Gooseff
159 et al., 2009), causing localised and strong influences on hydrochemical systems (Malone et al., 2013).
160 The effects of thermokarst are mostly local, but the transport of solutes into downstream aquatic
161 ecosystems can sometimes impact large areas (e.g., Kokelj et al., 2013). Moreover, the formation of new
162 features such as thaw ponds and lakes can modify the hydrochemical characteristics of large areas (e.g.,
163 Manasypov et al., 2014).

164 (iii) Rock-glacier thawing. Rock glaciers are distinctive and sometimes abundant geomorphological
165 features of cold mountain regions (Barsch, 1996). They are slowly-flowing mixtures of rock debris and ice

166 and formed by a continuum of processes from glacial to periglacial (Haeberli et al., 2006). The thawing of
167 ice-rich talus slopes (cf., Gruber and Haeberli, 2009), although not studied in detail, is likely part of the
168 same continuum. Rock glaciers are considered as the visible expression of long term creep of ice-rock
169 mixtures under permafrost conditions (Humlum, 2000; Ikeda and Matsuoka, 2006; Berthling, 2011), and
170 for this reason, they are often used for first-order mapping of permafrost distribution at coarse scales (e.g.,
171 Brenning, 2005; Cremonese et al., 2011; Schmid et al., 2015). Compared to glaciers, rock glaciers are
172 believed to react more slowly to rising temperature because of the retardation of heat transfer through
173 thick debris mantles (Barsch, 1996). During the cold season, ice is formed within rock glaciers, their “mass
174 balance”, however, is difficult to determine. Emerging observations suggest the thermal and kinematic
175 state of many rock glaciers is changing in concert with climate change (e.g., Roer et al., 2005; Haeberli et
176 al., 2006; Scapozza et al., 2014). Many active (moving) rock glaciers loose some water due to melt during
177 summer. This is because frequently, their tongues extent into microclimatic situations in which the moving
178 ice core undergoes slow degradation. However, at decadal and longer time scales, due to global warming,
179 accelerated melting of the ice within rock glaciers may represent an increasing hydrological contribution to
180 downstream areas (Brenning, 2005). For this reason, active rock glaciers are generally considered to be
181 sources of fresh water (Burger et al., 1999), especially in arid areas, small catchments, and during low
182 flow conditions (e.g., Corte, 1976; Brenning, 2005; Azócar and Brenning, 2010; Millar et al., 2013;
183 Rangecroft et al., 2015). Finally, melting of permafrost ice in active rock glaciers in particular, and also in
184 ice-rich talus (cf., Gruber and Haeberli 2009), is able to increase the porosity of the sediments. In turn, this
185 can increase the storage capacity, causing changes in the discharge pattern (Rogger et al., 2017).

186 The thaw of rock glaciers due to climate change has the potential to export enriched-solute fluxes and
187 thereby modify the inorganic chemistry of downstream impounded surface waters (e.g., Ilyashuk et al.,
188 2014) and streams (e.g., Thies et al., 2013).

189 Local and regional modifications of inorganic chemistry of surface fresh water following different
190 configurations of permafrost thaw and degradation have been reported globally and are listed in Tab. 1.
191 The impacts on major ions and trace elements are discussed in the following chapters.

192

193 4. Impacts on major ion content

194 4.1 Pervasive permafrost degradation

195 A strong divergence of inorganic solute concentrations between catchments with differing permafrost
196 extents has been reported, whereby lower concentrations are associated with higher permafrost extent.
197 This has been attributed to permafrost (a) inhibiting the infiltration of surface water into deep mineral soil
198 strata, and (b) confining mineral-rich groundwater in the subpermafrost zone without hydrological
199 connection to surface water (Frey and McClelland, 2009; Woo, 2012; Kane et al., 2013). For example,
200 Frey et al. (2007a) analysed Total Inorganic Solutes (TIS, defined as sum of Ca^{2+} , K^+ , Mg^{2+} , Na^+ , Si , Cl^- ,
201 SO_4^{2-} and HCO_3^-) as a function of latitude in West Siberian streams and rivers in Russia, showing that TIS
202 concentrations were 289 mg L^{-1} in permafrost-free watersheds compared to 48 mg L^{-1} on average in
203 watersheds with permafrost (Fig. 1). Other studies highlighted differences in concentrations of major ions,
204 such as HCO_3^- (Yukon River basin - Alaska, Striegl et al., 2005; Siberia - Russia, Yukon - Alaska, and
205 Mackenzie - Canada, Tank et al., 2012), Ca^{2+} and Mg^{2+} (Caribou-Poker Creeks Research Watershed -
206 Alaska, MacLean et al., 1999), Cl^- , SO_4^{2-} , Ca^{2+} , K^+ , Mg^{2+} , and Na^+ (Central Canadian sub-arctic - Canada,
207 Rühland et al., 2003), Na^+ , Ca^{2+} , Cl^- , and SO_4^{2-} (Central Mongolia, Szopińska et al., 2016), SO_4^{2-} , Ca^{2+} ,
208 Mg^{2+} , Na^+ , K^+ , and DIC (Dissolved Inorganic Carbon) (western Siberia - Russia, Pokrovsky et al., 2015),
209 among areas with different permafrost distribution. In western Siberia, Pokrovsky et al. (2015) also
210 provided evidence of the importance of plant litter and ground vegetation leaching as element sources
211 comparing K^+ concentrations as a function of latitude. Indeed, lower solute concentrations were measured
212 in permafrost-bearing zones in spring, during intense plant litter leaching, due to lower biomass and
213 primary productivity of forest-tundra and tundra biomes compared to the productive taiga zone of the
214 western Siberia. Based on their results, Pokrovsky et al. (2015) hypothesised that, assuming a short-term
215 climate warming scenario of hundreds of years, an increase in ALT will be capable to increase the export
216 of major elements from transitional permafrost watersheds.

217 Increasing contributions of highly-mineralised groundwater, and increased contact (time and volume)
218 between soil water and mineral surfaces have been proposed as the main drivers of ion enrichment in
219 surface water of several areas subject to permafrost degradation. A recent investigation performed by
220 Toohey et al. (2016) showed an increase in Ca^{2+} , Mg^{2+} , SO_4^{2-} , and Na^+ in the Yukon River Basin (Alaska)

221 in the period 1982-2014 (Fig. 2), attributing this evidence to altered hydrological flowpaths and increased
222 weathering due to widespread permafrost degradation. Several other studies of Northern-Hemisphere
223 high-latitude streams reported increasing concentrations of ions such as Ca^{2+} and SO_4^{2-} (Noatak National
224 Preserve - Alaska, Stottlemeyer, 2001), Ca^{2+} and Mg^{2+} (sub-Arctic - Sweden, Giesler et al., 2014), Ca^{2+} ,
225 Mg^{2+} , K^+ , and Na^+ (Caribou-Poker Creeks Research Watershed - Alaska, Petrone et al., 2006, 2007),
226 SO_4^{2-} , Ca^{2+} , Mg^{2+} , and alkalinity (Mackenzie basin - Canada, Tank et al., 2016), HCO_3^- , Cl^- , SO_4^{2-} , Ca^{2+} ,
227 and Mg^{2+} (Central Siberian Plateau - Russia, Kolosov et al., 2016), following permafrost degradation.
228 Similar findings have been reported for streams and lakes of the Colorado Rocky Mountains (USA) over
229 the last two-three decades, with increasing concentrations of Ca^{2+} and SO_4^{2-} (Baron et al., 2009; Caine,
230 2010; Todd et al., 2012), and Ca^{2+} , Mg^{2+} , and SO_4^{2-} (Mast et al., 2011).

231 Although increases in major ions concentrations of surface water have been found in several
232 transitional permafrost areas, the causal link with permafrost degradation is not well established. Keller et
233 al. (2007, 2010) and Douglas et al. (2013) proposed the use of Ca/Sr , Ca/Na , Ca/Ba , and $^{87}\text{Sr}/^{86}\text{Sr}$ trends
234 in soils and streams for quantitatively assessing the effects of increasing ALT on mineral fluxes to surface
235 water, where the parent material and soil geochemistry are variable with depth. For instance, geochemical
236 trends point to carbonate concentrations in the ground increasing with depth (higher concentrations in the
237 permafrost than in the active layer), due to decreasing frequency of thaw and hence weathering intensity.
238 Correspondingly, a progressive increase of carbonate weathering signals during the summer season
239 (especially in late summer) was found, associated with increasing thaw depth. New insights on
240 mechanisms and sources of inorganic solutes in surface waters draining permafrost-affected soils can
241 also be gained by using stable isotopes of inorganic solutes such as Ca (Bagard et al., 2013), Mg
242 (Mavromatis et al., 2014, 2016), and Si (Pokrovsky et al., 2013). For example, Pokrovsky et al. (2013)
243 analysed Si isotopic composition of large and small rivers, surface flow, interstitial soil solutions, plant litter
244 and soils in Central Siberia, finding significant variations of the dissolved Si isotopic composition. These
245 variations were linked to 1) the high concentrations of suspended silicate matter during the spring flood, 2)
246 the progressive increase of ALT during summer accompanied by secondary mineral precipitation in
247 surface layers and plant uptake of Si, and 3) the impact of the permafrost-hosted deep underground

248 waters during winter. Under climate change, the resulting permafrost degradation may be the most
249 important factor controlling the Si isotopic composition in the rivers.

250 Short-time increases in the depth of thaw are also capable to enhance mineral fluxes from permafrost
251 terrain. Increases in major ion concentrations (mainly Ca^{2+} , Mg^{2+} , Na^+ , SO_4^{2-} , and Cl^-) in surface water
252 have been linked to active-layer thickening following strong thermal perturbations (warm summer
253 conditions) and summer rainfall (Melville Island - Canada, Dugan et al., 2012, Lewis et al., 2012;
254 Lafrenière and Lamoureux, 2013; Roberts et al., 2017) with flushing of solutes stored in near-surface
255 permafrost (Melville Island - Canada, Lamhonwah et al., 2017) as well as to fire disturbance (Central
256 Siberia - Russia, Parham et al., 2013).

257 In the McMurdo Dry Valleys (Antarctica) the presence of unusual surface-flow-seep features has been
258 attributed to the potential melting of complex-origin sub-surface ice (e.g., excess ground ice, snow
259 patches, refrozen precipitation that has accumulated in the subsurface, or buried glacier ice) in permafrost
260 (Harris et al., 2007). Seeps showed a different geochemical signature in comparison to glacial meltwater,
261 strongly enriched in solutes (Na^+ , Ca^{2+} , Mg^{2+} , Cl^- , NO_3^- , SO_4^{2-} and HCO_3^-). These features associated with
262 water tracks (Levy et al., 2011) might contribute to the desert hydrological cycle exporting solute-
263 concentrated fluxes from sub-surface ice to surface water, enhancing the connectivity between the
264 cryosphere and the hydrosphere in the Antarctic polar desert (Levy et al., 2011).

265 Future increases in air temperature, and rain, may enhance the release of major ions from solute-rich
266 layers in near-surface permafrost. Indeed, several studies reported a solute enrichment at the base of the
267 active layer and at the top of permafrost in different permafrost environments due to ion redistribution
268 during thermally-induced moisture migration (Herschel Island - Canada, Kokelj et al., 2002; Northwest
269 Territories - Canada, Kokelj and Burn, 2003, 2005; Lacelle et al., 2014), greater availability of water for
270 chemical aqueous reactions where drainage is inhibited by subjacent permafrost (Yukon Territory -
271 Canada, Lacelle et al., 2007; Devon Island - Canada, Lacelle et al., 2008), mineral dissolution reactions
272 taking place in loamy layers slowly flushed by advective flow, with the reaction products transported
273 upward by diffusion to surficial peat layers frequently flushed during summer (Ilulissat - Greenland, Jessen
274 et al., 2014), physical modifications (landslides) of active layer geochemistry (Yamal Peninsula - Russia,

275 Leibman and Streletskaya, 1997), and cryogenic concentration of salts during freezing and humidity-
276 driven salt separation at the soil surface (McMurdo Dry Valleys - Antarctica, Levy et al. 2012).

277 Permafrost degradation can also increase the export of NO_3^- from soils to surface water due to deeper
278 pathways that bypass the shallow organic soil horizons where most retention and removal of inorganic
279 nitrogen occurs (Caribou Poker Creeks Research Watershed and Upper Kuparuk River Basin - Alaska,
280 Harms and Jones, 2012). For instance, increased nitrification and decreased nitrate uptake due to
281 enhanced water-inorganic soil and limited water-organic soil contacts have been demonstrated to lead to
282 higher nitrate concentrations in surface water in permafrost environments. Jones et al. (2005) investigated
283 three watersheds underlain by discontinuous permafrost in interior Alaska (Caribou Poker Creeks
284 Research Watershed) over a four-year period. The results showed that deeper flow paths in areas of less
285 extensive permafrost facilitated nitrate export, with the loss of permafrost that could also have released
286 stored nitrogen inaccessible to terrestrial plant uptake, resulting in high nitrification and leaching loss
287 rates. Petrone et al. (2006) followed up on this work and highlighted the sensitivity of nitrate fluxes to
288 changes in active layer thickness, with higher concentrations associated with deeper movements of water
289 into soils. The authors further demonstrated the role of the active layer as source of nitrate during the
290 summer season, especially during summer storms, as also reported by Koch et al. (2013) in the
291 Richardson Catchment (Alaska). Furthermore, McClelland et al. (2007) found a substantial increase in
292 nitrate export from the Upper Kuparuk River (Alaska) between the late 1970s and early 2000s, attributing
293 it to mechanisms potentially linked to warming and permafrost thaw.

294 Conversely, comparison of 96 rivers and streams in western Siberia found no differences in Dissolved
295 Inorganic Nitrogen (DIN) concentrations linked to variations in permafrost extent (Frey et al., 2007b). Frey
296 and McClelland (2009) attributed the difference in findings for western Siberia and Interior Alaska to the
297 greater remineralisation (and subsequent nitrification) of organic nitrogen in the Alaskan watersheds
298 (intermediate permafrost extent), in contrast with a limited remineralisation of organic nitrogen in western
299 Siberia due to limited soil water saturation in peat-dominated terrains and enhanced denitrification due to
300 wet conditions.

301 Deepening of flow paths can also lead to increasing concentrations of inorganic phosphate and silicate
302 in soil water due to enhanced mineral weathering, which is the primary source of these inorganic nutrients

303 (Frey and McClelland, 2009, Pokrovsky et al., 2015). For instance, Hobbie et al. (1999) demonstrated that
304 permafrost thawing along the MilkyWay stream in northern Alaska contributed 30% of inflowing phosphate
305 into Toolik Lake, causing bottom-up trophic effects. Similarly, increasing concentrations of dissolved
306 silicate with a decrease in permafrost coverage were found in Alaska (MacLean et al., 1999) and in
307 western Siberia (Frey et al., 2007a). However, assuming future climate warming and increasing vegetation
308 density in permafrost-bearing zones, transient uptake of nutrients during summer period may attenuate
309 the expected increase in surface water concentrations linked to progressive involvement of thawed
310 mineral horizons, as hypothesised by Pokrovsky et al. (2015) for Si dynamics in western Siberian rivers
311 (Russia).

312

313 **4.2 Thermokarst**

314 Thermokarst disturbances in permafrost terrain such as retrogressive thaw slumps, thermal erosion
315 gullies, active-layer detachment slides, and catastrophic thaw of frozen peat bogs have been found
316 to strongly modify major ion concentrations of surface water (mainly SO_4^{2-} , Ca^{2+} , Mg^{2+} , Na^+ , and Cl^-). The
317 concentrations of solutes released from thermokarst can be several orders of magnitude higher than in
318 background water, due to the liberation of soluble materials sequestered in ground ice. Such instances
319 have been reported globally, but mainly in Arctic areas: Ellesmere Island, Canada (Kokelj and Lewkowicz,
320 1999); Melville Island, Canada (Dugan et al., 2012); Banks Island, Canada (Rudy et al., 2017); Northwest
321 Territories, Canada (Kokelj et al., 2005; Malone et al., 2013); Mackenzie Delta, Canada (Kokelj et al.,
322 2009); Peel Plateau and Mackenzie Mountains, Canada (Kokelj et al., 2013); Brooks Range, Alaska
323 (Abbott et al., 2015); western Siberia, Russia (Loiko et al., 2017); and McMurdo Dry Valleys, Antarctica
324 (Gooseff et al., 2016).

325 Thermokarst can affect surface water chemistry for several decades with the magnitude of effects
326 depending on the proportion of a catchment area influenced by the disturbance (Kokelj et al., 2005) (Fig.
327 3), although generalisations are difficult since hydrochemical modifications can depend on several factors
328 such as the type of disturbance, its stabilisation rate, the downstream connectivity, and the characteristics
329 of the impacted area (e.g., physical and geochemical characteristics of near-surface permafrost,
330 mineralogical composition of bedrock/sediments).

331 Thermokarst processes can also release Si, and N and P forms into surface water. For example, Si
332 and P were found to be higher in small depressions and permafrost subsidences in actively thawing
333 frozen palsas peat bog areas than in stable palsas peat bog zones in western Siberia, Russia (Loiko et al.
334 2017). Moreover, thermokarst gullies and thaw slumps have been shown to increase N and P form
335 concentrations in water tracks, streams and lakes (Brooks Range - Alaska, Bowden et al., 2008; Harms et
336 al., 2014; McMurdo Dry Valleys - Antarctica, Gooseff et al., 2016). Louiseize et al. (2014) (Melville Island -
337 Canada) demonstrated that even small thermokarst occurrences (active-layer detachments) are capable
338 to promote the export of microbially-derived nitrate as result of limited nitrate retention and enhanced
339 nitrification in mineral soils exposed in the scar zones. Abbott et al. (2015) (Brooks Range - Alaska) also
340 suggested that aquatic disturbance of inorganic nitrogen by thermokarst can be long-lived. Conversely,
341 Thompson et al. (2012) (Mackenzie Delta uplands - Canada) found that lakes unaffected by thaw slumps
342 had higher concentrations of the nutrients than slump-affected lakes, despite the high concentrations of
343 inorganic N and P in runoff pools within the slump sites. Within-lake biological uptake, sedimentation and
344 degradation of nutrients were assumed to be the main drivers of this effect.

345

346 **4.3 Rock-glacier thawing**

347 Thies et al. (2007) found strong increases in electrical conductivity (up to 19-fold), SO_4^{2-} (up to 26-fold),
348 Ca^{2+} (up to 13-fold), and Mg^{2+} (up to 68-fold) from 1985 to 2005 in two lakes at the toe of active rock
349 glaciers in the Central Eastern Alps in Europe (Austria and Italy) (Fig. 4). These trends have been
350 attributed to the melting of ice in the rock glaciers and an associated release of solutes, although no direct
351 evidence of the driving processes was provided. A similar evidence was reported in 20 remote high-
352 elevation lakes located in Himalaya (Mt. Everest Region, Nepal) over the past two decades. Here,
353 significant enrichments in solutes have been attributed to the retreat of debris-covered glaciers (Salerno et
354 al., 2016) due to the sulphide oxidation processes that occur in subglacial environments, often microbially
355 mediated.

356 Increased geochemical fluxes from thawing permafrost and ice melt in rock glaciers have been also
357 found to influence the seasonal hydrochemistry of small streams in the Austrian Alps (Krainer and Mostler,
358 2002; Berger et al., 2004; Krainer et al., 2007), although no specific analyses on major ions were

359 conducted and instead, conductivity measurements were used as proxy. Williams et al. (2006) in 2003
360 filled this gap, reporting strong seasonal increases for SO_4^{2-} (60-fold), Mg^{2+} (30-fold) and Ca^{2+} (20-fold) in
361 the outflow of rock glacier RG5 in the Colorado Front Range (Rocky Mountains - USA) compared to
362 nearby streams and lakes during the low-flow season. Longer-term measurements from 1998 to 2002
363 showed the same pattern of elevated concentrations of these solutes in the fall. The elevated
364 concentrations of major ions in rock glacier outflows and downstream lakes are consistent with the results
365 of a precursor study performed by Giardino et al. (1992), who noted that water exiting rock glaciers in the
366 San Juan Mountains of Colorado (USA) acquired a significant dissolved load during movement through
367 the rock glaciers. Specific hydrochemical characteristics in rock glacier meltwaters were found also in the
368 hyperarid Agua Negra river basin - Andes of Argentina (meltwaters enriched in SO_4^{2-} , HCO_3^- and Ca^{2+} ,
369 Lecomte et al., 2008) and in Central Eastern Alps - Austria (meltwaters enriched in SO_4^{2-} , Ca^{2+} and Mg^{2+} ,
370 Thies et al., 2013).

371 During the last decades, the increasing solute concentrations in lakes with rock glaciers in their
372 catchments were attributed to the enhanced melting of ice due to atmospheric warming (Thies et al.,
373 2007). Williams et al. (2006) further supported this hypothesis, assuming this process may occur
374 seasonally, with a flushing of mineralised water at the end of summer when the 0 °C isotherm proceeds
375 towards the ice under conditions of degrading permafrost. Williams et al. (2006) reported the highest ion
376 concentrations in rock glacier outflows during particularly dry years. In this case, there would be more
377 internal melt than in wetter years, with a higher proportion of mineralised groundwater. The authors also
378 hypothesised that long-lasting snow cover might be able to reduce solute export by delaying subsurface
379 melt and by dilution of solute-rich water with snowmelt. Not only snowmelt, but also rain has been
380 reported to lower solute contents in rock glacier outflows (Krainer et al., 2007) through dilution effects.
381 Conversely, Thies et al. (2007) hypothesised that solute fluxes from rock glaciers intensify after periods of
382 intense rainfall during summer and fall, as these might enhance water percolation through rock glaciers
383 and thus cause the flushing of geochemical products. In summary, three hypotheses exist: (a) Warm
384 periods cause increased subsurface melt, which releases solutes. (b) Dry periods reduce the input of
385 meteoric water and thus reduce the dilution of rock-glacier outflow. (c) The percolation of meteoric water

386 through the rock glacier exports solutes. Considerable uncertainty exists about the processes, and their
387 interactions, driving the export of solutes from internal ice in rock glaciers.

388 Recent research attributed increased NO_3^- concentrations in surface water melt in permafrost and rock
389 glaciers (Rocky Mountains - USA, Baron et al., 2009). Williams et al. (2007) found that nitrate
390 concentrations in the outflow of a rock glacier were higher than all other analysed surface waters of the
391 same area in the Colorado Front Range, Rocky Mountains (USA), from 1998 to 2004 (Fig. 5). Fegel et al.
392 (2016) provides addition support that elevated biogeochemical and microbial characteristics of rock
393 glaciers in the Cascade Mountains, Rocky Mountains, and Sierra Nevada (USA) result in elevated nitrate
394 content in surface water. Further, microbial communities adapted to the extreme environment of the
395 interior of rock glaciers have been indicated as potential sources of the elevated nitrate in the rock glacier
396 outflows (Williams et al. 2007). The observed increase in surface-water nitrate concentration may also be
397 a result of melt water flushing microbially-active sediments (Barnes et al., 2014) and/or active microbial
398 populations in sediment pockets within talus fields in the Rocky Mountains (USA) (Ley et al., 2004).

399

400 **5. Impacts on trace elements**

401 **5.1 Pervasive permafrost degradation**

402 Although several studies dealt with the analysis of trace element concentrations in surface fresh water
403 in cold areas, especially in the boreal and sub-Arctic regions (e.g., Rember and Trefry, 2004; Voss et al.,
404 2015), researches devoted to the investigation of the links with permafrost dynamics are rare and mainly
405 based on short time series. Typically, mobilisation of trace elements to surface water during early spring is
406 attributed to snowmelt when precipitation that has accumulated during winter is released (McNamara et
407 al., 1997; Rember and Trefry, 2004; Pokrovsky et al., 2010), with trace elements also potentially
408 incorporated from upper-soil horizons and thawing ponds (Rember and Trefry, 2004). Differently, Barker
409 et al. (2014) reported increasing concentrations of trace elements (Al, Ba, Fe and Mn) in stream water
410 during late fall in Alaska (Brooks Range), correlated with increased ALT and weathering fluxes from
411 freshly-exposed mineral surface (metal sources). Bagard et al. (2011) discriminated two distinct seasonal
412 peaks of trace element fluxes in Central Siberia (Russia): the first one in the beginning of spring snowmelt
413 and related to the uppermost soil horizon rich in organics, and the second one occurring late in summer

414 and related to deeper mineral soil horizons. Permafrost gradient was found to be the most important
415 environmental parameter in controlling trace element concentrations in western Siberia (Russia), where
416 Pokrovsky et al. (2016) analysed approximately 60 large and small rivers (<100 to ≤150,000 km²
417 watershed area) during spring flood and summer, and winter baseflow, across a 1500-km latitudinal
418 gradient. The main factors controlling the shift of river feeding from surface and subsurface flow to deep
419 underground flow in the permafrost-bearing zone were the depth of the active layer and its position in
420 organic or mineral horizons of the soil profile. The authors also suggested that under climate warming
421 scenarios the change in the relative degree of the peat and mineral soil leaching to the river might cause a
422 decrease in divalent metal organic complexes and an increasing delivery of trivalent and tetravalent
423 hydrolysates in the form of organo-ferric colloids to the river via supra-permafrost flow and hyporheic
424 influx. Differently, a recent study performed by Raudina et al. (2017) in western Siberia (Russia) reported
425 that the concentration of most trace elements in peat porewater did not show any statistically significant
426 trend with latitude. Applying a space-for-time substitution approach for the climate warming scenario in
427 western Siberia, the authors also predicted a decrease in the feeding of small rivers and lakes by peat soil
428 leachates in transitional permafrost areas with frozen peatlands. Finally, similar to major elements, the use
429 of stable isotopes of inorganic solutes such as Fe (Hirst et al., 2017) will allow to improve the knowledge
430 about the impacts of future permafrost evolution on surface waters.

431 In a multi-decadal study (from 1980s to 2000s) performed in a mineralised alpine watershed with
432 natural acidic in the Colorado's Rocky Mountains (USA), Todd et al. (2012) provided evidence of
433 increasingly high concentrations (increase ranging from 100% to 400%) of Zn and other metals of
434 ecological concern (e.g., Mn) in the Upper Snake River during low-flow months (late summer-early fall)
435 (Fig. 6). Increased rock weathering on freshly-exposed mineral surfaces and sulphide oxidation due to the
436 degradation of permafrost and the opening of new subsurface pathways, causing the migration of
437 groundwater and oxygen into the subsurface, were indicated as driving factors. Acid drainage is indeed a
438 potential mechanism capable to strongly influence water quality, through the release of metals (e.g., Fe,
439 Zn, Pb, Cu) from sulphide-rich minerals due to the exposure to oxygen and water (e.g., He et al., 1997;
440 Nordstrom, 2011), in areas where permafrost is degrading. In northern Yukon Territory (Canada), Lacelle
441 et al. (2007) reported water quality deterioration due to acidity and high metal concentrations in a stream

442 affected by acid drainage. Here, acid drainage was a natural phenomenon which effects were amplified by
443 the construction of a road. The removal of the surficial sediments, and consequent deepening the active
444 layer, allowed seasonal thawing and oxidation of sulphide-rich minerals in the excavated area. Increase of
445 ALT under warmer climate scenarios might enhance acid drainage at sites with sulphide-rich minerals by
446 allowing access to atmospheric oxygen to penetrate deeper in the ground.

447

448 **5.2 Thermokarst**

449 Trace element dynamics in thermokarst ponds and lakes have been found to be primarily related to the
450 evolutionary stage of these features. For example, Manasypov et al. (2014) investigated the
451 hydrochemical characteristics of thermokarst ponds and lakes with different degrees of permafrost
452 influence located in the northern part of western Siberia, across a latitudinal gradient of ~900 km. The
453 authors evidenced a geographical gradient in trace element concentrations related to thermokarst lake
454 evolution (from small permafrost depressions to larger lakes), in relation to different degrees of carbon
455 and element leaching from frozen peat soils. They estimated that the acceleration of permafrost thaw in
456 Siberia might increase the amount of small thermokarst lakes and, in turn, increase the concentrations of
457 most trace elements by 200-400%. Shirokova et al. (2013) also reported a clear decrease in some trace
458 element concentrations (e.g., Fe, Al, Ti, Zr and rare earth elements) from thermokarst depressions, to
459 ponds, and finally, to lakes in the central part of western Siberia (Russia) (Fig. 7). They attributed this
460 pattern to two main factors: i) increase in water residence time in large lakes, which leads to more efficient
461 mineralisation of organo-mineral colloids by aerobic heterotrophic bacterioplankton; ii) decrease in peat
462 abrasion at the lake borders as the perimeter-volume ratio decreases with size. Furthermore, trace
463 element concentrations (e.g., Fe, Al, Ni, Pb) have been found to increase in hollows, depressions and
464 permafrost subsidences in an area characterised by thawing frozen peat in western Siberia, Russia (Loiko
465 et al., 2017). Here, the authors reported decreasing concentrations with increasing surface area of the
466 water body, along the hydrological continuum: soil water → hollows → depressions and permafrost
467 subsidences → thaw ponds → thermokarst lakes. Thermokarst lakes can also have an important role in
468 the storage of some trace elements (Al, Cr, Cd, Pb) comparable to, or even higher than, the transport of
469 these elements by rivers (western Siberia - Russia, Polishchuk et al., 2017). Hot summer periods, and

470 cycles of ice formation and melting have been reported to drive in-lake biological uptake and release of
471 some elements (Zn, Mn, Co, Mo, Cu, Si, Pb, Cd) (western Siberia - Russia, Pokrovsky et al., 2013), and
472 seasonal differences in trace element concentrations (western Siberia - Russia, Manasypov et al., 2015),
473 respectively. In addition, Audry et al. (2011) observed that in differently-developed thermokarst lakes in
474 western Siberia, sediments at the early stage of ecosystem development are a sink for water column Cu,
475 Zn, Cd, Pb and Sb due to authigenic sulphides precipitation, confirming the progressive depletion trends
476 of trace element concentrations in the water column during the chronosequence of lake maturation (from
477 permafrost subsidence to mature and drained lakes) reported by Pokrovsky et al. (2011), who analysed
478 20 lakes and small ponds of various sizes and ages in the same area. Conversely, the sediments are a
479 source of dissolved Co, Ni and As to the water column due to upward diffusive fluxes, regardless the
480 ecosystem development stage. However, the concentrations of these elements were found to be low due
481 to sorption processes on Fe-bounding particles and/or large-size organo-mineral colloids.

482 Several studies have been conducted focusing on Hg and Pb dynamics in thermokarst-affected areas.
483 For instance, Klaminder et al. (2008) assessed the impact of permafrost degradation on the increase in
484 Hg mobility from soils to lake surface water in a palsa peat complex in sub-Arctic Sweden. Main driving
485 factors were accelerated soil/peat erosion, increasing hydrological connectivity, and increased runoff
486 which led to the exposure of soluble elements. Gordon et al. (2016) (Northwest Territories - Canada)
487 further demonstrated that new hydrological pathways due to permafrost thaw can increase peatland
488 hydrological connectivity, creating areas of high Hg concentrations at critical points (fens) before water
489 discharged to a wider drainage system. Small permafrost thaw ponds also contribute in controlling the
490 local and regional fluxes of Hg, as demonstrated by MacMillan et al. (2015) in sub-Arctic and high-Arctic
491 Canada. Elevated concentrations of Hg and methylmercury have been found to be strongly related to
492 inputs of organic matter and nutrients into surface water. Moreover, Rydberg et al. (2010) reported that
493 long-term changes in climate, permafrost and mire dynamics had large effects on the release of Hg from
494 palsa mires to lakes in sub-arctic Sweden. Particularly, the recent thawing of permafrost has increased
495 thermokarst erosion, resulting in an enhanced Hg transport from the mire to lake water. In their study of
496 sub-Arctic Sweden, Klaminder et al. (2010) reported a similar process for Pb, attributing its increase to the
497 thermokarst erosion of peat. Conversely, Deison et al. (2012) found that lakes affected by the

498 development of retrogressive thaw slumps in the Mackenzie Delta (Canada) had lower Hg levels in
499 surface sediments when compared to not-affected lakes. The authors attributed these results to the
500 effects of thaw slumping of clay-rich permafrost tills, which resulted in high inorganic sedimentation rates
501 that caused a dilution of organic material and associated Hg in the sediments.

502

503 **5.3 Rock-glacier thawing**

504 An increase in trace element concentrations in high-elevation Alpine lakes (Central Eastern Alps,
505 Austria and Italy) has been observed, and it has been attributed to the influence of ice melting within rock
506 glaciers (Thies et al., 2007). For instance, unexpectedly high Ni, Mn and Al concentrations, exceeding the
507 European Union limits for drinking water by more than an order of magnitude, have been reported for a
508 remote high-mountain lake situated at the toe of an active rock glacier. In contrast, negligible
509 concentrations of these elements (and other solutes) have been recorded in an adjacent pond without a
510 rock glacier in the catchment (Central Eastern Alps - Italy, Ilyashuk et al., 2014). Surprisingly, the lithology
511 did not contain high Ni concentrations, thus the source of Ni remains unknown. In addition, other springs
512 derived from rock glaciers in the Central Eastern Alps exhibited high concentrations of trace elements
513 such as Ni, Co, Cu, Fe, Mn and Zn (Krainer, 2014), pointing to rock-glacier ice as a potential origin.
514 Increasing acidity of rock-glacier affected surface waters accompanied by high heavy metal
515 concentrations have been also reported to cause significant changes in aquatic ecosystems (Central
516 Eastern Alps - Austria, Thies et al., 2013) and severe ecological damages (Central Eastern Alps - Italy,
517 Ilyashuk et al., 2017).

518 Results from geochemical analyses of ice cores sampled from a medium-sized active rock glacier
519 (Lazaun) in Central Eastern Alps (Italy) are reported by Krainer et al. (2015). The ice near the base is
520 approximately 10,300 years old and trace elements (Ni, Co, Cu, Zn) are enriched in three horizons within
521 the ice at depths of 4.2 m (approx. 2600 yr BP), 9.9 m (approx. 3200y BP) and 12.4 m (approx. 3500 yr
522 BP), with Ni concentrations up to 0.49 mg L⁻¹. High Ni concentrations were also measured at the rock
523 glacier spring, indicating that Ni in spring water was derived from distinct depth intervals within the
524 permafrost ice (Krainer et al., 2011). Again, the lithology was reported to not contain high concentrations
525 of trace elements. However, only few rock glaciers in Central Eastern Alps display meltwater with

526 abnormally high trace element concentrations. Considerable uncertainty still exists about the origin of the
527 high concentrations of trace elements in rock-glacier ice.

528

529 **6. Research perspectives**

530 Several studies report major ion distribution in near-surface permafrost, finding increasing
531 concentrations at the top of permafrost, generally corresponding with ice-rich layers. However, relatively
532 few researches have combined geochemical analyses on changing near-surface permafrost and resulting
533 effects on surface-water characteristics. Performing combined investigations may provide insight into the
534 relationship between concentrations of major ions in near-surface permafrost and modifications of
535 surface-water geochemistry in case of permafrost degradation, both on intra- and inter-annual bases. Few
536 studies in the Arctic also demonstrated that rainfall events can flush solutes from near-surface permafrost
537 to the surface and increase ion concentrations in surface water. Projected warming and rainfall
538 augmentation in Arctic areas (IPCC, 2013) might cause increases in major ion concentrations in surface
539 water in permafrost areas; in turn, increasing runoff might reduce ion concentrations since discharge is
540 generally negatively correlated with concentrations of major ions because of dilution effects (Petroni et
541 al., 2006, 2007), although the “chemostatic” behaviour in stream water (solute concentrations in stream
542 water are not determined by simple dilution, Godsey et al., 2009) should be also taken into account. The
543 ability to adequately incorporate these processes into models will be relevant for predicting the
544 hydrochemical impacts of climatic changes in transitional permafrost areas. Future studies should also
545 focus on the bedrock/sediment composition and particularly on the geochemical composition of
546 permafrost ice that might be locally enriched in ions (due to lithological characteristics), which can be
547 released when permafrost ice melts. Finally, research in highly sensitive areas such as watersheds where
548 acid drainage is possible due to lithological properties (e.g., pyritiferous shale presence) will help to
549 understand how permafrost degradation may cause a deterioration of surface water quality.

550 Major ion concentrations downstream of rock glaciers have mainly been investigated in the European
551 Alps and in mountain chains of North America. Extending these studies to areas such as in the Andes
552 (e.g., Brenning, 2005) and in the Hindu Kush Himalaya (e.g., Schmid et al., 2015; Gruber et al., 2017) will
553 allow better understanding the dynamics and potential effects of rock-glacier thaw on downstream water

554 quality. The establishment of baselines for future monitoring and the better understanding of processes of
555 solute export from rock glaciers will thereby be important elements of such studies. In this framework,
556 investigating the hydrological connections between rock glaciers and adjacent surface waters through the
557 application on non-invasive techniques (Colombo et al., 2017) could provide new insights on the rock-
558 glacier role, and dynamics, in exporting concentrated chemical fluxes into high-elevation hydrological
559 systems. Moreover, it has been shown that melting of ice in active rock glaciers can increase their storage
560 capacity. May chemical composition of outflowing water be affected by this process? Fresh water stored in
561 rock glaciers can be important for local water management only during the melt season (summer) since
562 during the cold season (winter) discharge is extremely low or absent. Future studies on water quality
563 originating from rock glaciers should also focus on relict rock glaciers due to their higher storage capacity
564 and residence time for fresh water which can also be released during winter. Finally, extending
565 investigations to other common debris features with permafrost such as talus slopes (Sass, 2006; Lambiel
566 and Pieracci, 2008; Scapozza et al., 2011) will allow understanding the effects of thawing permafrost on
567 water quality in cold mountains more broadly. Do talus slopes with significant amount of ground ice (cf.,
568 Gruber and Haeberli, 2009) behave like rock glaciers in their export of solutes during permafrost
569 degradation? Trends of major ions and ion-isotope ratios in surface water may be considered as important
570 additions to the permafrost monitoring conducted in mountains (c.f., PERMOS, 2016).

571 Given that most studies on geochemistry in near-surface permafrost and ground ice considered only
572 major ions, future research on trace element concentrations and their variations within the active layer,
573 permafrost and massive ground ice occurrences is necessary for evaluating the effect they might have on
574 ecosystems when thawing.

575 High trace element concentrations have been reported downstream of rock glaciers and in three
576 horizons within the ice core of a rock glacier in the Central Eastern Alps. Since lithology was not found to
577 contain high trace element concentrations, alternative hypotheses (e.g., atmospheric fallout) to chemical
578 weathering should be tested in order to understand the origin and the potential for future release of trace
579 elements.

580 Beside specific research needs, several broader questions emerge for the understanding of permafrost
581 degradation and its effects on inorganic chemistry of surface water. These are especially important in arid

582 areas, such as some areas in the Andes and Himalaya, where fresh water stored in permafrost can be
583 important for local water management (Brenning, 2005; Rangecroft et al., 2013; Rangecroft et al., 2015;
584 Gruber et al., 2017) in small catchments. What is the role of permafrost in controlling water quality in these
585 areas and what duration will the transient effects of permafrost thaw have? What will be the long-term
586 implications of permafrost degradation on water quality of local freshwater reservoirs? What are the
587 possible socioeconomic consequences of water quality deterioration due to permafrost degradation? What
588 are suitable adaptation strategies to mitigate the possible impacts of permafrost degradation on water
589 supply quality of local populations?

590

591 **7. Conclusion**

592 This review of permafrost-thaw impacts on inorganic chemistry of surface fresh water has revealed
593 several common patterns.

594 Spatially-distributed progressive increases of major ion delivery to surface fresh water have been
595 reported in both polar and mountain areas following permafrost degradation. This is the result of
596 increasing contributions of highly mineralised groundwater and enhanced interactions of soil water with
597 deep mineral strata. Localised releases of major ions to surface water due to the liberation of soluble
598 materials sequestered in permafrost and ground ice have been found in ice-rich terrains both at high
599 latitude (thermokarst features) and high elevation (rock glaciers). Generalisations must be considered
600 carefully though, because the intensity of hydrochemical modifications depends on several factors such as
601 the characteristics of the watersheds (e.g., physical and geochemical characteristics of permafrost), their
602 hydrological connectivity, the magnitude and type of disturbance in case of thermokarst, and the extent of
603 permafrost or rock glaciers in the catchment. Permafrost degradation can also increase the export of
604 inorganic N and P forms, and Si, from permafrost to surface water due to deeper flow paths that bypass
605 shallow organic layers where most retention and removal of nutrients occur (N), and because of enhanced
606 mineral weathering (P and Si). However, significant uncertainty has been shown on DIN dynamics, mainly
607 dependent on regional and landscape-scale differences in Arctic areas (e.g., Alaska and Siberia). In
608 addition, thermokarst can enhance the export of inorganic nutrients from permafrost to surface water,
609 although biological uptake, sedimentation and degradation of nutrients might mitigate these effects.

610 Increases in nitrate concentrations have been also attributed to rock glacier melt water flushing microbially
611 active sediments.

612 Trace element dynamics in permafrost areas have been shown to exhibit two seasonal peaks in
613 geochemical fluxes: one at the beginning of the snowmelt (early spring), related to the flushing of surficial
614 soil horizons and previously mobilised species stored in them; the second occurring in late summer and
615 early fall, associated with deeper mineral-soil horizons. The formation of new thermokarst features (e.g.,
616 ponds, lakes, palsa mires) and leaching from frozen peat can be relevant for increasing trace element
617 concentrations in permafrost catchments and for affecting the locations of high concentrations due to
618 newly-developing hydrological pathways. Alternatively, increasing sediment erosion due to thermokarst
619 processes has been reported to potentially mitigate these effects through increasing sedimentation rates
620 and dilution of concentrations. Moreover, stabilisation and increasing size of older thermokarst features
621 might lead to decreasing trends because of stabilising shorelines, reduced peat abrasion, and more
622 efficient mineralisation. Increasing trends in trace element concentrations have been also reported in
623 streams and lakes in mountain areas, where enhanced chemical weathering on fresh mineral surfaces
624 exposed by permafrost thaw and contribution of metal-enriched groundwater have been hypothesised to
625 be the main drivers. Nevertheless, in the Central Eastern Alps, high concentrations of trace elements
626 (e.g., Ni) in surface water have been found to be unrelated to the lithological setting, hence rejecting the
627 hypothesis of mineral weathering as the only process involved.

628 In conclusion, several factors influence the timing and magnitude of the impacts of permafrost
629 degradation on inorganic chemistry of surface fresh water. These include permafrost sensitivity to thawing
630 (e.g., amount of ground ice), modes of permafrost degradation (e.g., increasing ALT, thermokarst) and
631 environmental characteristics of watersheds (e.g., soil properties, lithology, land cover, topography,
632 hydrological connectivity). Further mobilisation of solutes and related transport to surface fresh water
633 appear reasonable under warming climatic conditions. This might develop through gradual concentration
634 increase in case of progressive and diffused thawing of permafrost, and/or through distinct temporal peaks
635 in case of localised thermokarst and rock glacier thawing. As a result, modifications of inorganic water
636 chemistry might have significant implications for ecological and human systems.

637

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643

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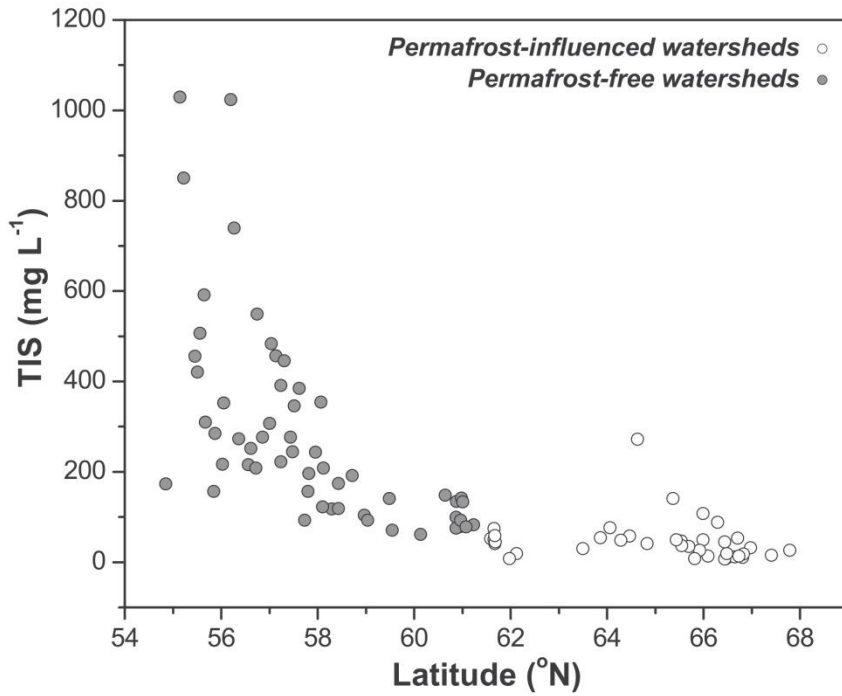
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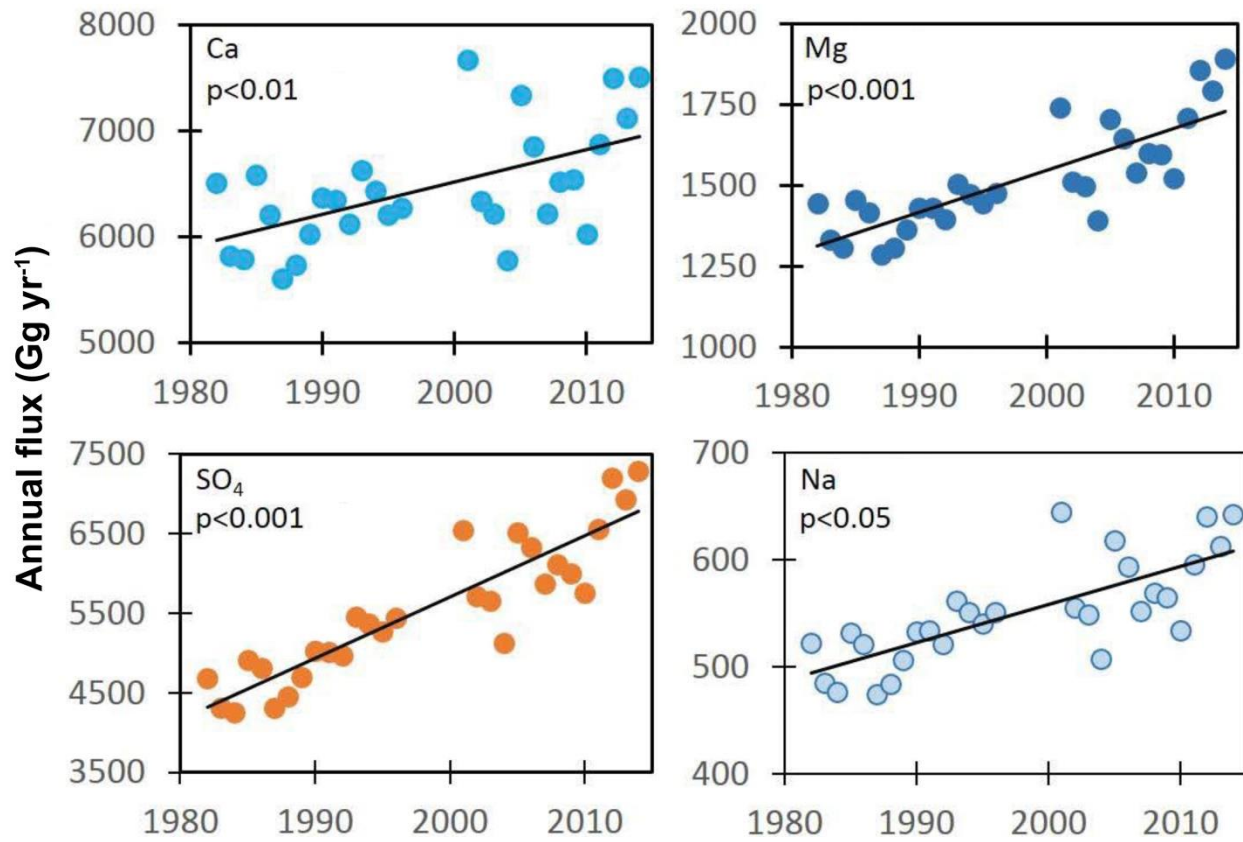
1134 **Figures and Tables**



1135

1136 Figure 1 - TIS (total inorganic solutes, defined as sum of Ca²⁺, K⁺, Mg²⁺, Na⁺, Si, Cl⁻, SO₄²⁻ and HCO₃⁻) as

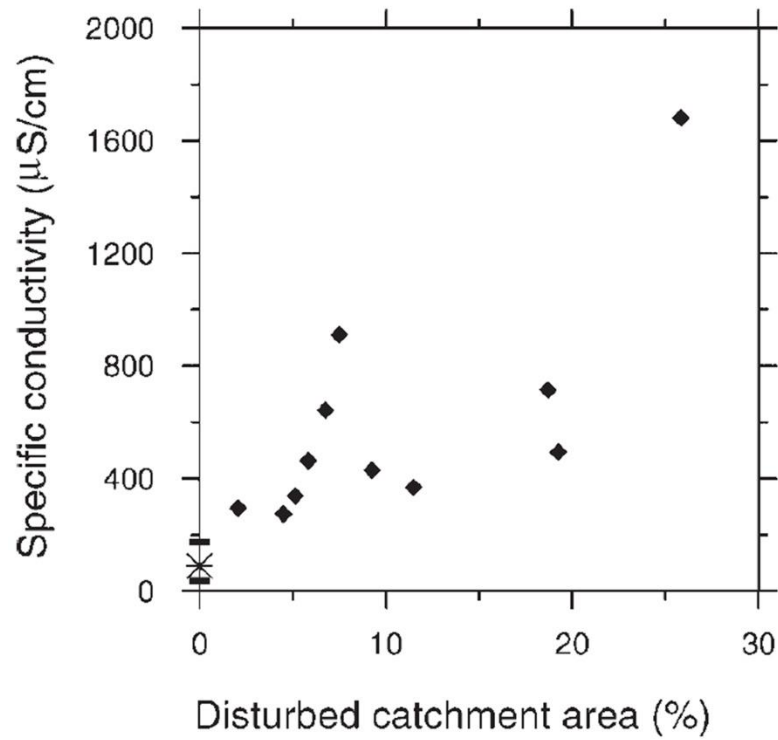
1137 a function of latitude. Permafrost limit is approximately located at 61°N (from Frey et al., 2007a).



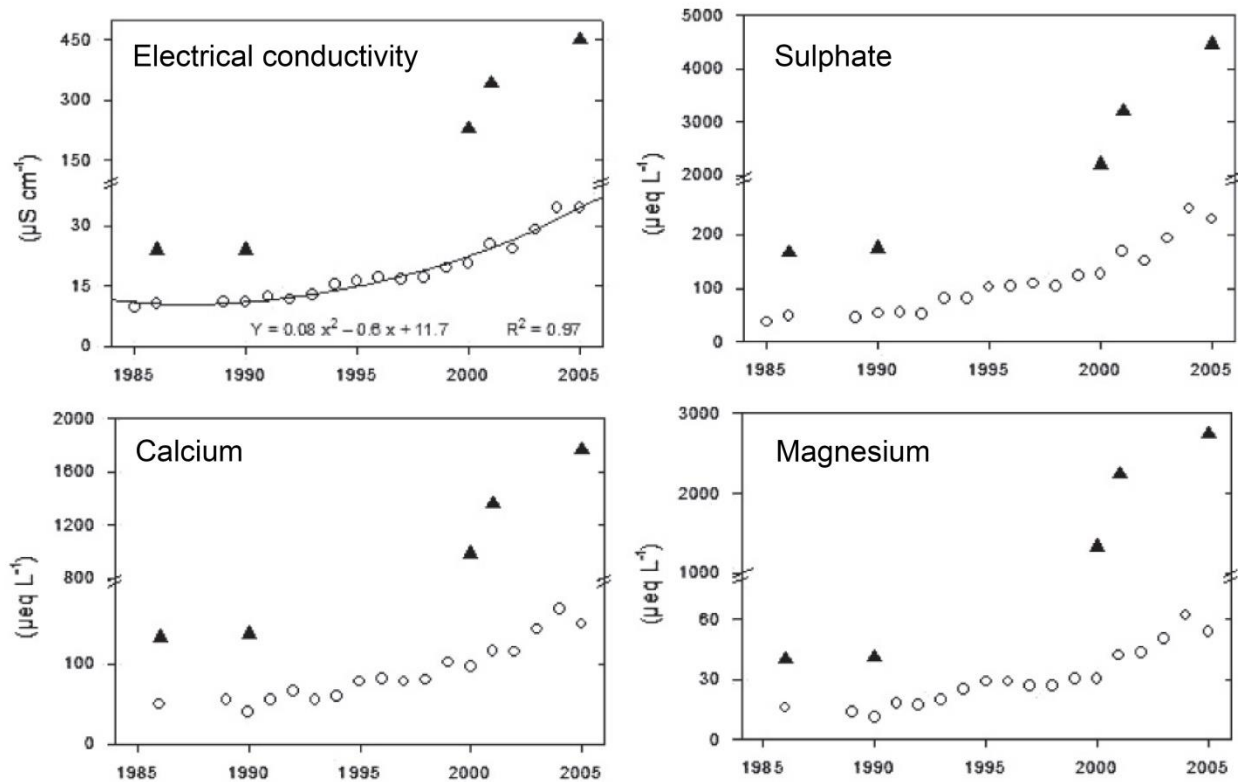
1138

1139 Figure 2 - Annual flux increases in the Yukon River with Thiel-Sen trend lines (adapted from Toohy et al.,

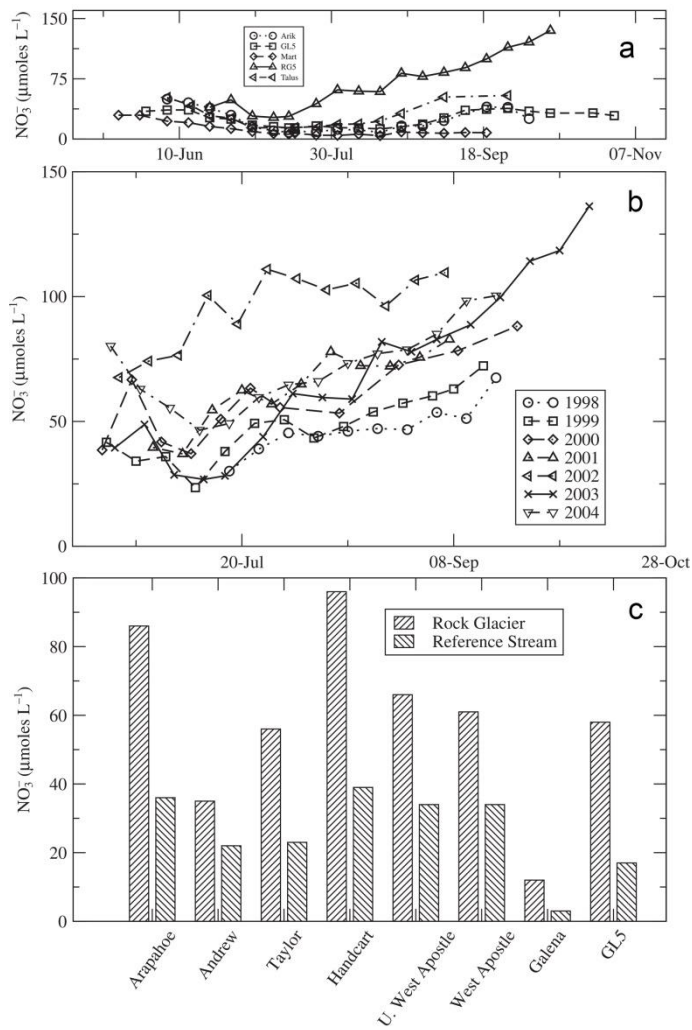
1140 2016).



1141
 1142 Figure 3 - Percentage of catchment area influenced by thermokarst disturbance (slumping) and specific
 1143 conductivity (as index of total solute concentrations, mainly represented by Ca^{2+} , Mg^{2+} , and SO_4^{2-}) of lake
 1144 water, upland lakes, Mackenzie Delta region. The (*) indicates the mean specific conductivity of the 11
 1145 pristine lakes measured by Kokelj et al. (2005). The (-) indicates maximum and minimum values for
 1146 pristine lakes (from Kokelj et al., 2005).

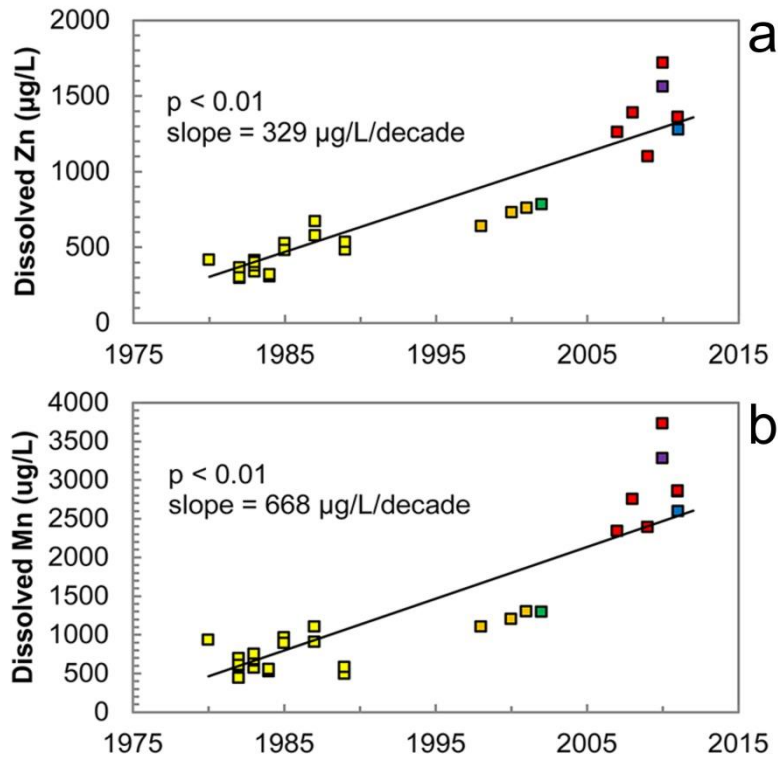


1147
 1148 Figure 4 - Electrical conductivity, sulphate, calcium and magnesium concentrations in lake water of
 1149 Rasass See (black triangles) and Schwarzsee ob Sölden (open circles) lakes (1985-2005). Values for
 1150 each lake represent mean values of four to seven discrete samples along the lake vertical profile taken
 1151 during holomixis. Variability among single values is <5 %. Horizontal lines show break in vertical scale
 1152 (adapted from Thies et al., 2007).

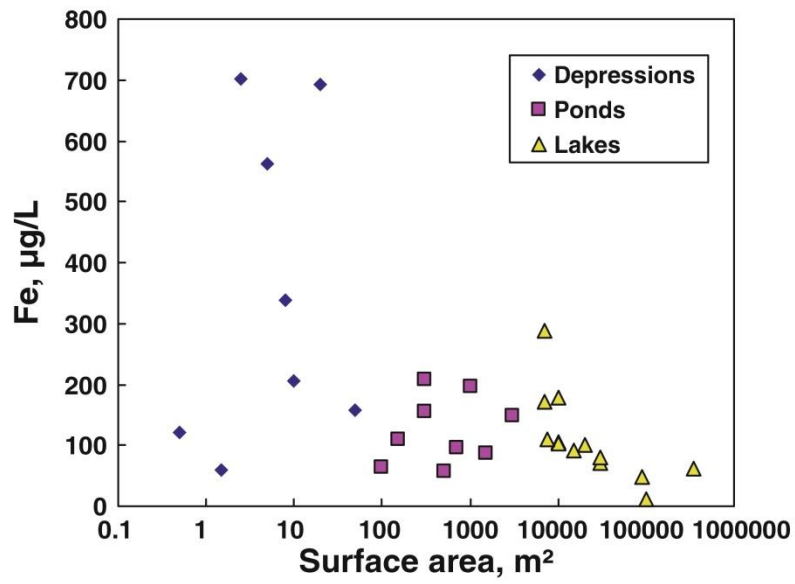


1153
 1154 Figure 5 - a) Time series of nitrate in surface water of the Green Lakes Valley, for 2003. Arik is Arikaree
 1155 Glacier outflow; GL5 is the outlet of Green Lakes 5; Mart is the outlet of the Martinelli catchment; RG5 is
 1156 the Green Lake 5 rock glacier outflow; talus is the blockfield site (for site details refer to Williams et al.,
 1157 2007). b) Time series of nitrate concentrations in the outflow of RG5 during the snow-free season from
 1158 1998 to 2004. c) Comparison of nitrate concentrations from the outflow of 7 rock glaciers and co-located
 1159 surface waters, August 2003 (adapted from Williams et al., 2007).

1160



1161
 1162 Figure 6 - Trends in September dissolved Zn (a) and Mn (b) concentrations (µg/L) in the Upper Snake
 1163 River. Different colours represent different data sources (further details in Todd et al., 2012). p-values
 1164 were computed using the Mann-Kendall Test, and lines shown are Kendall-Theil lines (adapted from Todd
 1165 et al., 2012).



1166

1167 Figure 7 - Iron (<math><0.45 \mu\text{m}</math>) concentration as a function of water body surface area in depressions, ponds

1168 and lakes. Uncertainties ($\pm 2\sigma$) are the same or smaller than symbol sizes (from Shirokova et al., 2013).

Author	Year	Study area	Permafrost configuration	Analytes
Bagard et al.	2013	Central Siberia - Russia	PD	M
Caine	2010	Colorado Rocky Mountains - USA	PD	M
Douglas et al.	2013	Chena River - Alaska	PD	M
Frey et al.	2007a	Western Siberia - Russia	PD	M
	2007b			
Giesler et al.	2014	Sub-Arctic - Sweden	PD	M
Harms and Jones	2012	Caribou Poker Creeks Research Watershed, Upper Kuparuk River Basin - Alaska	PD	M
Harris et al.	2007	McMurdo Dry Valleys - Antarctica	PD	M
Hobbie et al.	1999	MilkyWay stream - Alaska	PD	M
Jones et al.	2005	Caribou Poker Creeks Research Watershed - Alaska	PD	M
Keller et al.	2007	Brooks Range - Alaska	PD	M
	2010			
Koch et al.	2013	Richardson Catchment - Alaska	PD	M
Kolosov et al.	2016	Central Siberian Plateau - Russia	PD	M
Lafrenière and Lamoureux	2013	Melville Island - Canada	PD	M
Lamhonwah et al.	2017	Melville Island - Canada	PD	M
Levy et al.	2011	McMurdo Dry Valleys - Antarctica	PD	M
Lewis et al.	2012	Melville Island - Canada	PD	M
MacLean et al.	1999	Caribou-Poker Creeks Research Watershed - Alaska	PD	M
Mast et al.	2011	Colorado Rocky Mountains - USA	PD	M
Mavromatis et al.	2014	Central Siberia - Russia	PD	M
	2016	Yenisey River - Russia		
McClelland et al.	2007	Upper Kuparuk River - Alaska	PD	M
Parham et al.	2013	Central Siberia - Russia	PD	M
Petrone et al.	2006	Caribou-Poker Creeks Research Watershed - Alaska	PD	M
	2007			
Pokrovsky et al.	2013	Central Siberia - Russia	PD	M
	2015	Western Siberia - Russia		
Roberts et al.	2017	Melville Island - Canada	PD	M
Rühland et al.	2003	Central Canadian sub-Arctic	PD	M
Stottlemeyer	2001	Noatak National Preserve - Alaska	PD	M
Striegl et al.	2005	Yukon River basin - Alaska	PD	M
Szopińska et al.	2016	Central Mongolia	PD	M
Tank et al.	2012	Siberia - Russia, Yukon - Alaska,	PD	M
	2016	Mackenzie - Canada		
Todd et al.	2012	Colorado Rocky Mountains - USA	PD	M / T
Dugan et al.	2012	Melville Island - Canada	PD / TK	M
Baron et al.	2009	Colorado Rocky Mountains - USA	PD / RG	M
Abbott et al.	2015	Brooks Range - Alaska	TK	M
Bowden et al.	2008	Brooks Range - Alaska	TK	M
Gooseff et al.	2016	McMurdo Dry Valleys -Antarctica	TK	M
Harms et al.	2014	Brooks Range - Alaska	TK	M
Kokelj and Lewkowicz	1999	Ellesmere Island - Canada	TK	M
Kokelj et al.	2005	Northwest Territories - Canada	TK	M
Kokelj et al.	2009	Mackenzie Delta - Canada		

Kokelj et al.	2013	Peel Plateau, Mackenzie Mountains - Canada		
Loiko et al.	2017	Western Siberia - Russia	TK	M / T
Louiseize et al.	2014	Melville Island - Canada	TK	M
Malone et al.	2013	Northwest Territories - Canada	TK	M
Rudy et al.	2017	Banks Island - Canada	TK	M
Thompson et al.	2012	Mackenzie Delta uplands - Canada	TK	M
Barnes et al.	2014	Colorado Rocky Mountains - USA	RG	M
Berger et al.	2004	Central Eastern Alps - Austria	RG	M
Fegel et al.	2016	Cascade Mountains, Rocky Mountains, Sierra Nevada - USA	RG	M
Giardino et al.	1992	San Juan Mountains of Colorado - USA	RG	M
Krainer and Mostler	2002	Central Eastern Alps - Austria	RG	M
Krainer et al.	2007	Central Eastern Alps - Austria	RG	M
Lecomte et al.	2008	Agua Negra river basin - Andes of Argentina	RG	M
Thies et al.	2007	Central Eastern Alps - Austria, Italy	RG	M / T
	2013	Central Eastern Alps - Austria		
Williams et al.	2006	Colorado Front Range - USA	RG	M
	2007			
Bagard et al.	2011	Central Siberia - Russia	PD	T
Barker et al.	2014	Brooks Range - Alaska	PD	T
Hirst et al.	2017	Lena River Basin - Russia	PD	T
Lacelle et al.	2007	Northern Yukon Territory - Canada	PD	T
Pokrovsky et al.	2016	Western Siberia - Russia	PD	T
Raudina et al.	2017	Western Siberia - Russia	PD	T
Audry et al.	2011	Western Siberia - Russia	TK	T
Deison et al.	2012	Mackenzie Delta - Canada	TK	T
Gordon et al.	2016	Northwest Territories - Canada	TK	T
Klaminder et al.	2008	Sub-Arctic - Sweden	TK	T
	2010			
MacMillan et al.	2015	Sub-arctic, high-Arctic - Canada	TK	T
Manasypov et al.	2014	Western Siberia - Russia	TK	T
	2015			
Pokrovsky et al.	2011	Western Siberia - Russia	TK	T
	2013			
Polishchuk et al.	2017	Western Siberia - Russia	TK	T
Rydberg et al.	2010	Sub-Arctic - Sweden	TK	T
Shirokova et al.	2013	Western Siberia - Russia	TK	T
Ilyashuk et al.	2014	Central Eastern Alps - Italy	RG	T
	2017			
Krainer	2014	Central Eastern Alps - Austria, Italy	RG	T
Krainer et al.	2011	Central Eastern Alps - Italy	RG	T

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1170 Table 1 - Selected papers reporting information on the impacts of permafrost degradation on inorganic

1171 chemistry of surface fresh water globally. PD: Pervasive permafrost degradation, TK: Thermokarst, RG:

1172 Rock-glacier thawing, M: Major ions, T: Trace elements.