

1 **Effects of permafrost thaw on arctic aquatic ecosystems**

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Abstract

The Arctic is a water-rich region, with freshwater systems covering 16% of the northern permafrost landscape. Permafrost thaw creates new freshwater ecosystems, while at the same time modifying the existing lakes, streams, and rivers that are impacted by thaw. Here, we describe the current state of knowledge regarding how permafrost thaw affects lentic (still) and lotic (moving) systems, exploring the effects of both thermokarst (thawing and collapse of ice-rich permafrost) and deepening of the active layer (the surface soil layer that thaws and refreezes each year). Within thermokarst, we further differentiate between the effects of thermokarst in lowland areas, versus that on hillslopes. For almost all of the processes that we explore, the effects of thaw vary regionally, and between lake and stream systems. Much of this regional variation is caused by differences in ground ice content, topography, soil type, and permafrost coverage. Together, these modifying factors determine (i) the degree to which permafrost thaw manifests as thermokarst, (ii) whether thermokarst leads to slumping or the formation of thermokarst lakes, and (iii) the manner in which constituent delivery to freshwater systems is altered by thaw. Differences in thaw-enabled constituent delivery can be considerable, with these modifying factors determining, for example, the balance between delivery of particulate versus dissolved constituents, and inorganic versus organic materials. Changes in the composition of thaw-impacted waters, coupled with changes in lake morphology, can strongly affect the physical and optical properties of thermokarst lakes. The ecology of thaw-impacted lakes and streams, is also likely to change, these systems have unique microbiological communities, and show differences in respiration, primary production, and food web structure that are largely driven by differences in sediment, dissolved organic matter, and nutrient delivery. The degree to which thaw enables the delivery of dissolved versus particulate organic matter, coupled with the composition of that organic matter and the morphology and stratification characteristics of recipient systems will play an important role in determining the balance between the release of organic matter as greenhouse gases (CO₂ and CH₄), its burial in sediments, and its loss downstream. The magnitude of thaw impacts on northern aquatic ecosystems is increasing, as is the prevalence of thaw-impacted lakes and streams. There is therefore an urgent need to address the key gaps in understanding in order to predict the full effects of permafrost thaw on aquatic ecosystems throughout the Arctic, and their consequential feedbacks to climate.

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1. Introduction

Permafrost is perennially frozen ground that underlies about a quarter of the landmass of the Northern hemisphere (Brown et al., 1998). It consists of various soil types ranging from frozen peat and frozen mineral soil to frozen Pleistocene deposits rich in massive ground ice. The distribution of permafrost is generally divided into four zones based on the percentage of the land that is underlain by permafrost: continuous (90-100%), discontinuous (50-90%), sporadic (10-50%), and isolated (<10%) (Fig. 1). Terrestrial permafrost (we do not consider subsea permafrost in this review) hosts about 1,330-1,580 Pg (1Pg = 10¹⁵ g) organic carbon (OC) of which about half is deeper than 1 m (Schuur et al., 2015). Over the last few decades, permafrost ground temperatures

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82 have been slowly increasing (Romanovsky et al., 2010) as a result of increased surface warming in
83 Arctic regions (IPCC, 2013).

84 The Arctic is extremely rich in water. Lakes, reservoirs, rivers, and various types of wetlands,
85 floodplains, bogs, fens, and mires, on average occupy 16% of the landscape underlain by permafrost
86 when considering water bodies larger than 0.1 km² in area (Fig. 1; Global Lakes and Wetlands
87 Database; Lehner and Döll, 2004) making this number a conservative estimate. Discontinuous,
88 sporadic, and isolated permafrost regions are relatively rich in surface water with 20, 23, and 18%
89 landscape coverage, respectively, compared to continuous permafrost with only 11%. Water, in all
90 its forms, connects all components of the landscape and plays a key role in the storage and
91 transport pathways of sediments, organic matter, nutrients, and other constituents (Battin et al.,
92 2009; Vonk and Gustafsson, 2013). The role of hydrology is therefore key in both the response and
93 the effects of permafrost thaw, and it strongly influences the balance of carbon dioxide (CO₂) and
94 methane (CH₄) emissions. At the same time, permafrost thaw will also create new aquatic
95 ecosystems and modify existing aquatic ecosystems. In this review we provide an overview of the
96 effects of permafrost thaw on aquatic ecosystems and their potential feedbacks to climate, with a
97 consideration of all aquatic ecosystems located within the permafrost zones defined by Brown et al.
98 (1998; Fig. 1).

99 When permafrost thaws, the soil organic matter and minerals within it become available for
100 remobilization and introduction into aquatic systems. The type of thaw will largely determine the
101 rate and effects of this remobilization. Here, we will distinguish between two types of thaw: (i) thaw
102 of ice-rich permafrost, also called thermokarst, with a more abrupt or episodic character that tends
103 to manifest as a *pulse disturbance*, and (ii) thaw of permafrost with (relatively) low ground-ice
104 content, with a gradual but persistent and longer-term character that tends to manifest as a *press*
105 *disturbance* (Grosse et al., 2011). In this review we consider the effects of both pulse and press thaw
106 disturbance.

107 Thaw of ice-rich permafrost leads to a range of landscape features (Fig. 2) that are collectively
108 referred to as thermokarst, and that are typically divided into three primary groups based on where
109 and how they form (Kokelj and Jorgenson, 2013): (i) hillslope processes, (ii) wetland processes, and
110 (iii) thermokarst lake processes. *Hillslope processes* result in dramatic features such as
111 retrogressive thaw slumps (Fig. 2b), active layer detachment slides, and thermal erosional gullies
112 (Fig. 2c), which together we refer to as thermo-erosional features (TEFs). The scale of these
113 features is local and depends on landscape features, but the transport of sediment, nutrients, and
114 organic matter into aquatic ecosystems can be large, and these features may form in a matter of
115 hours and slowly grow for several years. *Wetland processes* include peatland collapse and the
116 development of bogs and fens. While wetlands store significant quantities of soil carbon (Tarnocai
117 et al., 2009) they are not specifically discussed in this review. *Thermokarst lake processes*, which
118 include lake and pond formation, expansion, and drainage, are the most abundant (10-50% of
119 permafrost-impacted landscapes; Jorgenson et al., 2006; Kokelj and Jorgenson, 2013) and most
120 easily recognizable form of thermokarst, and are particularly emphasized in this review.

121 Thaw of permafrost with lower ice content results in a more gradual top-down thawing process,
122 and occurs through active layer deepening and talik formation (Schuur et al., 2008). This type of

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thaw is gradual but can occur over entire landscapes (Åkerman and Johansson, 2008; Shiklomanov et al., 2013). Although its impact on aquatic ecosystems is harder to detect and requires long-term (decades-scale) monitoring, such gradual thaw can cause striking changes to regional hydrology and chemistry (e.g., in Alaska, Striegl et al., 2005; Keller et al., 2010; Walvoord et al., 2012). Talik formation occurs under water bodies, but also when the active layer has deepened to such an extent that the soil does not refreeze completely in winter. Taliks create new hydrological hotspots in the landscape, and may increase groundwater flow and deeper flow paths, particularly in the case of open taliks that fully penetrate the permafrost profile (Walvoord et al., 2012), and alter fluxes of constituents (Walvoord and Striegl, 2007).

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While ground ice content and topography are important modifying factors that determine how permafrost thaw manifests itself within landscapes, factors such as the local composition of soils and regional extent of permafrost will also play an important role in determining the effect of permafrost thaw on aquatic ecosystems (Fig. 3). For example, continued permafrost degradation, and in particular the transition from continuous to discontinuous permafrost, will considerably lengthen flow paths, often enable greater inputs of groundwater to freshwater systems, and affect the processing of water en route to aquatic systems (e.g., Striegl et al., 2005; Walvoord et al., 2012). Thermokarst processes along with other climate-driven changes in, for example, vegetation and precipitation, also determine local hydrological trajectories that affect the landscape in contrasting ways (e.g., lake expansion vs. drainage) (Bouchard et al., 2013; Turner et al., 2014). Similarly, local soil conditions (e.g., composition, porosity) will affect how permafrost thaw changes soil-water interactions as water moves across landscapes, with the mobilization or exposure of organic versus mineral soils being an important potential regulator of how thaw affects impacted aquatic systems (Tank et al., 2012a).

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In this review, we first describe the general impacts of permafrost thaw on aquatic ecosystems such as changes in physical, optical, and chemical limnology and the release of materials from land to water (section 2). We identify the various pathways of organic carbon and contaminant degradation (bio- and photodegradation), and the resulting changes in microbial community structure and gas fluxes from thaw-impacted systems (section 3). We then evaluate the broader consequences of permafrost thaw for aquatic ecosystems considering the release of old carbon into waters and the atmosphere, carbon burial, effects for ecosystem structure and functioning, and exports to the ocean (section 4). We end this review with a summary of our findings, an overview of potential climate feedbacks and recommendations for future research (section 5).

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2. Impacts of thaw on aquatic ecosystems

2.1 Physical and optical limnology

The land area above latitude 45.5°N totals 41.3 million km² and contains around 200,000 lakes (sized 0.1 to 50 km²), with 73% of them occurring in permafrost landscapes (Smith et al., 2007; Grosse et al., 2011; 2013). Thermokarst lakes and ponds, defined as water bodies that form in a depression as a result of permafrost thaw, are among the most abundant water bodies in permafrost landscapes, and can be found throughout the circumpolar north, from North America to

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Europe and Russia. They encompass a wide range of physical characteristics, which in turn contribute to the large variations in their biogeochemical properties. In this section, we focus on the effects of thermokarst on the physical and optical limnology of ponds and lakes (Fig. 4). We acknowledge, however, that thermokarst processes will also have significant effects on the physical properties of stream and river systems, for example via the delivery of coloured or chromophoric dissolved organic matter (CDOM) and sediments to these environments (see section 2.2).

Thermokarst lakes occur in cold, high-latitude environments that experience prolonged winters with sub-zero temperatures for 8 months or more of the year, and short summers with air temperatures that may rise well above 10°C depending on location. They tend to have minimal hydrological connectivity given their frozen surroundings, which inhibit the infiltration and exchange of water. As a result, the hydrological balance of these closed basins is strongly influenced by snowmelt in spring, evaporation and precipitation in summer, and water inflow from local permafrost soils as a result of thermokarst processes (Fig. 4; Dingman et al., 1980; Bowling et al., 2003; Smith et al., 2007). Notable exceptions are the thermokarst lakes located in floodplain deltas, where river waters may flood and connect the lakes each year, depending on their height above the main stem of the river (McKnight et al., 2008). Thermokarst lake disappearance is a common process throughout the Arctic (e.g., in Siberia, Smith et al., 2005; Kravtsova and Bystrova, 2009).

Thermokarst lakes vary greatly in surface area, from ponds that are only a few meters across (northern Québec, Breton et al., 2009; Bouchard et al., 2011) to lakes that are several kilometres in their maximum dimension (Alaska, Pelletier, 2005 and Arp et al., 2011; western Siberia, Pokrovsky et al., 2011). In some thermokarst-impacted landscapes, ponds and lakes can cover up to 30% of land surface area (estimated with remote sensing in NW Canada, Côté and Burn, 2002; and in Alaska, Hinkel et al., 2005), although subpixel-scale water bodies could substantially increase the total water surface area as was demonstrated in the Lena Delta of Siberia (Muster et al., 2013). Their depth is limited by local geomorphology and thaw depth of the permafrost, resulting in shallow waters in some areas, and much deeper systems in other regions. Depth is an important feature in determining lake characteristics as it determines whether water bodies are frozen to the bottom in winter. For thermokarst lakes in northern Québec, in a region extending from continuous to discontinuous and sporadic permafrost, maximum lake depths range from 1 to 3.5 m (Breton et al., 2009; Laurion et al., 2010; Bouchard et al., 2011; Crevecoeur et al., 2015). Similarly, lake depths range from 0.5 to 1.5 m on a discontinuous permafrost tundra in western Siberia (Pokrovsky et al., 2013), from 0.4 to 2.6 m in an area of continuous permafrost in northern Alaska (Arp et al., 2011), and from 1 to 3.5 m in continuous permafrost of the Arctic Coastal Plain (Hinkel et al., 2012). Thermokarst lakes approaching 10 m depth can be found in interior Alaska (Sepulveda-Jáuregui et al., 2015) and lakes deeper than 10 m can be found on the Seward Peninsula, Alaska (Hopkins, 1949) and northeast Siberia (Walter Anthony and Anthony, 2013); thermokarst lakes as deep as 22 m exist on the Yukon Coastal Plain of north-western Canada (West and Plug, 2008). The size and abundance of thermokarst lakes are changing, but with pronounced differences between regions. In Russia, thermokarst lake area has been increasing in mid-latitudes but decreasing in southern Siberia (Kirpotin et al., 2008; Sharonov et al., 2012). However, High Arctic thermokarst lakes have also been expanding (e.g. on the Yamal peninsula, Sannikov et al., 2012). Lake area changes may vary substantially (i.e., both increasing and decreasing) even within smaller regions, for example, in

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Altai (Polishuk et al., 2013) or northwest Siberia (Bryksina et al., 2011). In summary, thermokarst lakes and ponds occur in wide varieties (Fig. 2) depending on, for example, their formation process, surface area, and depth; from here on we generally distinguish between ponds and lakes based on their depth (ponds freeze to the bottom in winter, lakes do not), but also sometimes use 'lake' loosely for both systems when the distinction is less clear, or we follow the terminology used in the studies we cite.

Ongoing thermokarst can result in variable inputs of dissolved organic carbon (DOC), inorganic solutes (including nutrients and major ions), and particulate organic and mineral materials into these lakes. Consequently, their limnological properties are highly influenced by their regional surroundings (Fig. 4; Prowse et al., 2006; Bowden et al., 2008; Watanabe et al., 2011; see section 2.2 below). One of the expressions of that variability is the colour of the water, which although typically brown, may sometimes be blue, green, black or even white. A study by Watanabe et al. (2011) on water colour and light attenuation in the thaw waters of Nunavik, Québec (Fig. 2h), showed that the lake surface colour (water-leaving spectral radiance) was dependent upon the combined concentrations of CDOM and suspended non-algal particulate material, allowing certain biogeochemical properties such as DOC to be estimated from satellite remote sensing. Waters rich in CDOM (or in clays and silts) strongly attenuate solar radiation, and suppress primary production (see section 4.3.1). In shallower and clearer waters, sufficient photosynthetically-active radiation may penetrate to the bottom of the lakes to allow the development of aquatic macrophytes; for example in lakes of the Mackenzie River Delta that are less affected by flooding with turbid river water (Squires and Lesack, 2003).

Depending on their CDOM and particle content, the surface waters of thermokarst lakes may strongly absorb solar radiation, and this can give rise to pronounced surface warming of the more coloured lakes. In combination with the cooling of their bottom waters by the permafrost beneath, this means that some thermokarst lakes have pronounced vertical thermal and density gradients in summer (Fig. 4, Alaska, Sepulveda-Jáuregui et al., 2015). In winter, ice-cover reduces transfer of energy with the atmosphere and eliminates potential wind mixing, although some transfer of heat may result from the oxidation of organic matter in bacterial sediment processes if lakes are not frozen to the bottom (Mortimer and Mackereth, 1958).

Many thermokarst lakes are likely to be cold polymictic, undergoing stratification events that become established and then break down over diurnal, or several day, cycles (Alaska, Hinkel et al., 2012; Canadian sub-Arctic, Deshpande et al., 2015). The increased use of high resolution, automated temperature loggers is likely to yield new insights into these short term stratification and mixing dynamics, even in those lakes currently considered to be well-mixed in summer. For example, in western Siberia, an air-temperature increase of 15°C from an anomalous heat wave resulted in increased surface-water temperatures of 10°C, and the formation of a strong temperature gradient in the water column (Pokrovsky et al., 2013). Strong thermal stratification during summer has been reported in shallow (<3 m maximum depth) thermokarst lakes and ponds in sub-Arctic Québec (Laurion et al., 2010; Deshpande et al., 2015), the high Canadian Arctic (Bouchard et al., 2015)), northern Siberia (Boike et al., 2015), and in lakes on yedoma-like permafrost in a latitudinal transect in Alaska (Sepulveda-Jáuregui et al., 2015). In these shallow,

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297 stratified systems, oxygen depletion is more prevalent especially in bottom waters (Deshpande et
 298 al., 2015; Sepulveda-Jáuregui et al., 2015). On the other hand, lakes in non-yedoma permafrost or
 299 non-permafrost catchments in Alaska tended to be less stratified, with well-oxygenated bottom
 300 waters. In thermokarst lakes located at higher latitudes, particularly in tundra zones with reduced
 301 vegetation, little or no stratification has been observed during summer (Yukon territory, Canada,
 302 Burn, 2003; Alaska, Hinkel et al., 2012; western Siberia, Pokrovsky et al., 2013), possibly as a result
 303 of greater wind exposure (a function of fetch and wind speed), the reduced tendency of cold waters
 304 to stratify (the change in water density per unit °C is lower at lower temperatures), increased
 305 convective mixing, and less near-surface light absorption and heating. Deep thermokarst lakes (>2
 306 m; not freezing to the bottom in winter) show strong inverse stratification beneath their ice-cover
 307 during winter (e.g., Canadian (sub-)Arctic, Breton et al., 2009; Laurion et al., 2010; Deshpande et al.,
 308 2015; Alaska, Sepulveda-Jáuregui et al., 2015; Siberia, Boike et al., 2015), including those lakes that
 309 show little or no stratification in summer (e.g., Yukon territory, Canada, Burn, 2003). Shallow water
 310 bodies (depth <2 m) may freeze all the way to the bottom in winter; for example on the Arctic
 311 Coastal Plain of Alaska, where the ice thickness reaches 1.5 – 2 m depths (Arp et al., 2011).

312 Stratification within lakes is also influenced by gradients in salinity and gas concentrations that
 313 contribute to density differences down the water column (Kirillin et al., 2012). Such effects are also
 314 likely in thermokarst lakes, but have received little attention to date. Thermokarst processes result
 315 in inputs of eroded permafrost material into these water bodies, including POC and DOC, nutrients,
 316 and ions (Mackenzie delta uplands, Kokelj et al., 2005; circum-Arctic, Prowse et al., 2006; Alaska,
 317 Bowden et al., 2008), which are further concentrated with the lake freeze-up during winter
 318 (Canadian Arctic, Grasby et al., 2013; Canadian sub-Arctic, Deshpande et al., 2015), in turn affecting
 319 water column stability. The hydrodynamic effects of high dissolved CH₄ and CO₂ gradients under ice
 320 in stratified thermokarst lakes and of the gas bubble trains associated with ebullition from
 321 sediments (Siberia, Walter et al., 2006) have received little attention.

322 The combination of high rates of bacterial metabolism, small lake volumes, and prolonged ice cover
 323 means that thermokarst lakes (that do not freeze to the bottom in winter) can experience full water
 324 column anoxia for much of the year, in striking contrast to the well-known deeper, non-
 325 thermokarst lakes in the Arctic such as Toolik Lake, Alaska, and Char Lake, Canada (Deshpande et
 326 al., 2015). In sub-Arctic Québec thermokarst lakes, mixing in spring occurs during an extremely
 327 short period of time (< 5 d) before the lakes restratify (Laurion et al., 2010). Mixing at that time
 328 may be insufficient to completely re-oxygenate the water column, while in fall, prolonged mixing,
 329 likely aided by convective processes, results in the transfer of oxygen to the bottom of the lake
 330 (Deshpande et al., 2015). In a study near Mayo, Yukon Territory (Canada) by Burn (2003), a similar
 331 pattern was observed, of prolonged, substantial mixing in fall but only a short period of mixing in
 332 spring, which may favour the continuation of bottom water anoxia throughout summer. The fall
 333 mixing period is likely to be especially important for gas exchange with the atmosphere, and for
 334 stimulating aerobic processes such as bacterial respiration and methanotrophy throughout the
 335 water column.

336 Presently, there are major gaps in our understanding of the physical and hydrological dynamics of
 337 thermokarst lakes, including measurements of heat transfer from the sediments, the penetration of

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395 solar radiation through winter ice cover, the potential for internal seiches in winter influenced by
 396 floating ice, and the nature of groundwater flows (Kirillin et al., 2012). Advective transfer of liquid
 397 water from shallower littoral zones to the pelagic bottom waters due to differential cooling may
 398 also play a role in material transfer within these water bodies, as has been observed elsewhere
 399 (MacIntyre and Melack, 1995); no studies to date have addressed the three dimensional
 400 hydrodynamics of thaw waters.

401 2.2 Chemical limnology and the transfer of materials from land to water

402 Permafrost thaw and thermokarst processes can have a major effect on the chemistry of impacted
 403 aquatic ecosystems. Particularly in thermokarst lakes, these changes may occur as a result of the
 404 optical and physical processes described elsewhere. Where thermokarst affects lake stratification,
 405 for example, changes in phosphorus remobilization from sediments could occur (e.g., Sondergaard
 406 et al., 2003). However these effects have been little studied in thaw-impacted systems. Similarly,
 407 changes in water column and sediment oxygenation will change the prevalence of redox reactions,
 408 including the bacterially mediated processes described in sections 3.4 and 3.5, below. In addition to
 409 these classic 'limnological' changes, permafrost thaw can also be expected to have a fundamental
 410 effect on the transfer of sediments and chemical constituents from land to water. A recent review
 411 by Frey and McClelland (2009) provides an in-depth discussion of how this change may play out for
 412 nutrients, DOC, and major ions in stream and river systems. In this section, we update this previous
 413 work, and add a consideration of the specific effects of thermokarst and permafrost thaw via active
 414 layer deepening on lake water chemistry.

415 2.2.1 Press vs. pulse disturbances

416 The degree to which nutrients, organic matter, and sediments are released to aquatic systems is
 417 likely to depend on the type of permafrost thaw. For example, the press disturbance of active layer
 418 deepening (time scale of decades, and greater) will likely favour the delivery of soluble materials
 419 (nutrients, base cations, DOC), although the mechanisms that deliver soluble materials to aquatic
 420 systems are complex, as outlined below (see 2.2.3). In contrast, the pulse disturbance of thermo-
 421 erosional processes (localized time scale of years to decades) is likely to favour the delivery of
 422 particulate over soluble materials. In addition, hydrologic connectivity and landscape topography
 423 are also likely to affect land-to-water constituent transfer (Abnizova et al. 2014), and further affect
 424 carbon burial or transfer to the atmosphere. For example, high-gradient watersheds may
 425 experience much more lateral constituent transfer, while in low-gradient watersheds with low
 426 specific runoff the vertical emission of carbon as CO₂ and CH₄ may predominate.

427 The spatial distribution and life cycle of press and pulse disturbances will also govern the impact
 428 that they have on the delivery of biogeochemical constituents to aquatic ecosystems. For example,
 429 TEFs (thermo-erosional features) are discretely distributed across the landscape, following
 430 variations in topography (affecting, for example, snow cover; Godin et al., 2015) and ground ice
 431 content. While these features can be numerous in impacted areas (Lacelle et al., 2015) they take up
 432 a relatively small percentage of the total landscape area (1.5% in Alaska; Krieger, 2012). Individual
 433 TEFs have lifecycles on the order of decades (Kokelj et al., 2013; Pearce et al., 2014), and – while
 434 they are active – may have intense local impacts on sediment and ionic fluxes to freshwater systems

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(see below). [They also seem likely to act as a population of features, however, only a small portion of which will be active at any one time within the landscape. As a result, the observed significant local effects](#), may become more muted when averaged over wider landscapes. In contrast, the [press disturbance](#) of active layer thickening occurs more universally across the landscape, but results in changes that are much more subtle, and may require long-term [\(decades-scale\)](#) monitoring programs to detect. Slow press changes can operate over [many](#) decades, altering aquatic ecosystems (e.g., [in Alaska](#), Keller et al., 2010, Walvoord et al., 2012), and causing entire landscapes to slowly subside ([Alaska](#), Shiklomanov et al., 2013).

2.2.2 Sediment delivery to aquatic ecosystems

Thermo-erosional features either directly adjacent to, or within the catchment of, aquatic systems can significantly increase suspended sediment concentrations, particularly in streams and rivers where turbulence causes materials to remain entrained. Thaw slumps and gullies directly adjacent to streams have been shown to cause order-of-magnitude increases in suspended sediment concentrations ([Alaska](#), Bowden et al., 2008; Calhoun, 2012) that can continue to be seen for considerable distance downstream ([western Canadian Arctic](#), Kokelj et al., 2013). For example, one small thermokarst gully that formed in 2003 and intersected a small, headwater beaded-stream (the Toolik River) in a 0.9 km² Alaskan catchment delivered more sediment downslope to the river than is normally delivered in 18 years from a 132 km² adjacent reference catchment of the upper Kuparuk River (Bowden et al., 2008). Similarly, streams impacted by within-catchment active layer detachments have been shown to exhibit elevated sediment levels at their outflow when compared to non-impacted sites ([Canadian High Arctic](#), Lamoureux and Lafrenière, 2014). Where slumping occurs directly adjacent to lakes, however, slump-associated sediments can settle out of suspension rapidly, [depending on particle size](#). Thus, while lakes impacted by permafrost slumping experience altered sedimentation rates ([Mackenzie delta uplands](#), Deison et al., 2012), water column sediment loads are generally not impacted ([Mackenzie delta uplands](#), Kokelj et al., 2005; [Canadian High Arctic](#), Dugan et al., 2012). Similarly, increases in sediment loads are atypical in systems where thermokarst causes landscape collapse without significant exposure of soils, such as in the creation of thermokarst lakes in lowland regions. [However, where postglacial silts and clays are present \(which have very low sedimentation rates\), for example on the Eastern coast of Hudson Bay, permafrost thaw can have profound consequences on sediment loads to lakes \(Bouchard et al., 2011; Watanabe et al., 2011\).](#)

2.2.3 Organic matter delivery to aquatic ecosystems

[Increases](#) in sediment delivery to aquatic ecosystems will [in turn increase](#) the flux of particulate organic carbon (POC) to [affected](#) systems. In the active-layer detachment system described above, increases in suspended sediments were accompanied by measured increases in POC ([Canadian High Arctic](#), Lamoureux and Lafrenière, 2014). In lakes, permafrost thaw can change the rate of accumulation of organic matter in sediments, and depending on the composition of eroding materials, either increase ([sub-Arctic Sweden](#), Vonk et al., 2012a) or decrease ([Mackenzie delta uplands](#), Deison et al., 2012) the concentration of sediment organic matter. Notably, the POC that travels to aquatic systems as a result of permafrost thaw may be only partially derived from

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516 permafrost carbon, because the action of thaw and landscape collapse will also expose and mobilize
 517 soils from the seasonally unfrozen active layer (e.g., Kokelj and Jorgenson, 2013). In a later section
 518 (4.1.2), we review the effect of permafrost thaw on the mobilization of old organic carbon.

519 Where permafrost thaw enhances contact between water and organic soil horizons, increases in
 520 DOC concentrations are likely to occur. Direct slumping of old, yedoma carbon into streams causes
 521 striking increases in DOC in adjacent receiving waters in the Kolyma River watershed (Vonk et al.,
 522 2013), while slumping adjacent to streams in the Alaskan Arctic is also associated with significantly
 523 increased [stream water](#) DOC at the site of impact (Abbott et al., 2014). During thaw of ice-rich
 524 permafrost, DOC stored in ice wedges and other ground ice (Fritz et al., 2015) is also released.
 525 Thermokarst lakes that form in organic-rich terrains can have significantly elevated concentrations
 526 of DOC as a result of direct contact between overlying water and recently submerged soils, and
 527 continued thermokarst expansion into new soils at the lake margin (e.g., [sub-Arctic Québec](#), Breton
 528 et al., 2009; [western Siberia](#), [Shirokova et al., 2013](#)). In regions where slumping increases delivery
 529 of inorganic particles from land to water, however, aquatic DOC concentrations can decrease, as a
 530 result of the adsorption of organics onto sediment surfaces that settle after suspension (e.g.,
 531 [Mackenzie delta uplands](#), Kokelj et al., 2005; [sub-Arctic Québec](#), [Bouchard et al., 2012](#)).

532 In addition, thaw-enabled changes in [flow paths](#) can also be expected to affect the transport of DOC
 533 to aquatic ecosystems. Although there is little direct evidence for the effect of water interactions
 534 with deeper soil layers as active layers deepen in organic-rich regions, there are parallels to be
 535 drawn with more transitional (sub-[Arctic](#)) systems, where permafrost peatland plateaus are
 536 associated with low annual export (2–3 g C/m²/yr) dominated by the snow melt period (~70%)
 537 and non-permafrost fens are characterized by much higher DOC export (7 g C/m²/yr) due to more
 538 sustained annual hydrological connectivity ([sub-Arctic Sweden](#), Olefeldt and Roulet, 2014).
 539 Conversely, where soils are characterized by shallow organic layers, growing season export of flow-
 540 weighted DOC has been shown to decrease significantly [between 1978–1980 and 2001–2003](#)
 541 ([Yukon River, Alaska](#), Striegl et al., 2005), likely as a result of the combined effect of increased flow
 542 paths (deeper active layer), residence time ([Alaska](#), Koch et al., 2013), and microbial mineralization
 543 of DOC in the unfrozen soil and groundwater zone. Permafrost thaw as a result of wildfire has been
 544 shown to increase hydrologic connectivity between burned hillslopes and catchment surface
 545 waters, such that burned soils can become a dominant source of water and solutes to streams
 546 during summer, whereas unburned hillslopes provide longer term storage of water and solutes
 547 ([Alaska](#); Koch et al., 2014). [Recent forest fires in central Siberia \(Parham et al., 2013\) however, led](#)
 548 [to a decrease in stream DOC concentrations due to removal of a DOC source through combustion.](#) In
 549 these regions, it has been suggested that organic matter sorption onto newly thawed ([due to forest](#)
 550 [fires](#)) mineral soils may also be important (Petrone et al., 2007). Over geographic gradients,
 551 changes in permafrost extent [appear to](#) have regionally-variable effects on DOC flux from land to
 552 water, with decreasing permafrost extent (and presumably increasing contact with deeper soils and
 553 groundwater inflows) causing increasing DOC fluxes in organic rich regions, but decreasing DOC
 554 fluxes in regions with poorly developed organic horizons ([Frey and Smith, 2005](#); [Prokushkin et al.](#)
 555 [2011](#); [Tank et al., 2012a](#)). Controlled leaching experiments of soils from the Alaskan and western
 556 Canadian Arctic have also found that regardless of temperature and leaching time, only small

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566 | amounts of DOC are released from permafrost-impacted soils, and that mobilization of OC occurred
 567 | largely in the POC phase (Guo et al., 2007).

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568 | 2.2.4 Nutrient delivery to aquatic ecosystems

569 | Similar to DOC, the effect of permafrost thaw on nutrient concentrations may also be region, or
 570 | landscape specific. For thermo-erosional processes, direct slumping into Alaskan streams has been
 571 | shown to increase dissolved inorganic and organic nutrient concentrations (Bowden et al., 2008;
 572 | Abbott et al., 2014), while in the western Canadian Arctic, total dissolved N and P can be lower in
 573 | shallow lakes directly impacted by permafrost slumping. This latter effect has been hypothesized to
 574 | occur as a result of the adsorption of organic nitrogen and phosphorus onto settling mineral
 575 | particles, or following rapid uptake and senescence, and then burial, of photosynthesizing cells
 576 | (Mackenzie delta uplands, Thompson et al., 2012). In lakes that remain turbid following
 577 | thermokarst disturbances, total phosphorus concentrations can be high, following phosphorus
 578 | adsorption onto clays that are transported into aquatic systems (sub-Arctic Québec, Breton et al.,
 579 | 2009). Similarly, slumping can also increase sediment nutrient concentrations (Mackenzie delta
 580 | uplands, Mesquita et al., 2010), while shoreline expansion of thermokarst lakes in yedoma regions
 581 | can enable nutrient-rich yedoma soils, and the nutrient-rich plants that these soils support, to enter
 582 | lakes (Siberia, Walter Anthony et al., 2014).

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583 | Warming, coupled with landscape changes that decrease water contact with organic soils and
 584 | increase water contact with inorganic soils has been shown in several Arctic regions to lead to
 585 | higher nitrate concentrations in adjacent streams, as a result of decreased NO₃ uptake or increased
 586 | nitrification (e.g., Alaska, Jones et al., 2005 and Harms and Jones, 2012; Canadian High Arctic,
 587 | Louiseize et al., 2014). On the Alaskan North Slope, nitrate export from the upper Kuparuk River
 588 | increased over a period of several decades, via mechanisms that may be linked to warming and
 589 | permafrost thaw (McClelland et al., 2007). Conversely, deeper flow paths that increase contact with
 590 | mineral soils are expected to decrease dissolved organic nitrogen exports (Alaska, Walvoord and
 591 | Striegl, 2007; Harms and Jones, 2012; Koch et al., 2013). This may also lead to increased
 592 | phosphorus concentrations, because mineral weathering is the primary source of phosphorus in
 593 | soil waters (e.g., Frey and McClelland, 2009).

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594 | 2.2.5 Delivery of major ions

595 | Permafrost thaw is also expected to increase the concentration of weathering-derived ions in
 596 | receiving waters, as slumping or deepening flow paths increases the contact between water and
 597 | deeper mineral soil layers (see review in Frey and McClelland, 2009). For example, in the Alaskan
 598 | Arctic, increasing thaw depths have been associated with increasing surface water concentrations
 599 | of calcium and bicarbonate (Keller et al., 2010), while near-surface permafrost has been found to be
 600 | ion-rich in the western Canadian Arctic. In Siberia, and at multiple locations throughout the pan-
 601 | Arctic, decreasing permafrost extent has also been associated with increasing stream water
 602 | concentrations of major weathering ions (Frey et al., 2007; Tank et al., 2012a). This effect can be
 603 | pronounced when thermokarst slumping occurs directly adjacent to aquatic systems: both streams
 604 | (NW Canada, Malone et al., 2013) and lakes (Mackenzie delta uplands, Kokelj et al., 2005) can
 605 | exhibit strikingly elevated ionic concentrations when directly impacted by permafrost slumping.

2.2.6 Mobilization of contaminants

Contaminants reach polar regions following long-range atmospheric transport and deposition, as well as through increased local waste production from marine transport and industrial or mining activities. The frozen soils of permafrost have historically been considered a barrier to the movement of contaminants and many waste and dump sites use containment strategies that rely on the low mobility of contaminants in permafrost soils (Grannas et al., 2013). However, the warming Arctic climate may lead to an increased mobility of contaminants, either stored in soils at waste sites or historically-accumulated in permafrost, into Arctic surface waters (Armitage and Wania, 2013; Chételat et al., 2014).

Thawing permafrost may have major consequences for contaminant transport and transformations in the Arctic due to: a) physical changes in the hydrological cycle leading to the remobilization of contaminants from contaminated soils or sediments, b) chemical changes due to the release of nutrients and organic carbon from previously frozen soils, and c) biological changes via the microbial transformation of contaminants. Most studies on the interactions between contaminants and permafrost soils have concentrated on fuel products, persistent organic pollutants (POPs), and metals such as mercury (Hg) and lead (Pb).

Permafrost thaw may cause increased mobility of contaminants from catchment soils to surface waters due to accelerated soil/peat erosion, altered hydrological flow (increasing hydrological connectivity), and increased runoff leading to exposure of soluble contaminants (sub-Arctic Sweden, Klaminder et al., 2008). Increased lateral hydraulic conductivity may accelerate the downhill movement of contaminants through the large pores, lenses, and veins created in the active layer by the thawing of ice-rich permafrost (Mackenzie delta region, Dyke, 2001). After thaw, permafrost is no longer an impermeable barrier to contaminants, allowing for infiltration into soils and aquatic systems (Grannas et al., 2013). The reduced surface area of thermokarst lakes in some areas due to the creation of drainage channels may lead to increased contaminant concentrations in the remaining surface waters (Macdonald et al., 2005).

Traditionally, the distinction is made between (i) organic contaminants, and (ii) inorganic contaminants. Studies of *organic contaminants* (hydrocarbons or non-aqueous phase liquids) predict lateral movement in the active layer with limited vertical transport in areas of continuous permafrost (Mackenzie delta region, Dyke, 2001; Alaska, Carlson and Barnes, 2011). Vertical migration is possible in some regions (Alaska, McCarthy et al., 2004) yet permafrost acts as a low-permeability barrier in others (Antarctica, Curtosi et al., 2007). Organic contaminants may migrate downwards into frozen soils in areas of discontinuous permafrost due to more variable distribution (Alaska, Carlson and Barnes, 2011).

Studies of *inorganic contaminants* show that physical changes in permafrost affect contaminant mobility. For instance, Manasypov et al. (2014) observed geographical gradients in trace metal concentrations in surface waters from Siberia related to thermokarst lake evolution (from small permafrost depressions to larger lakes), mainly due to peat leaching. They also described seasonal differences in trace metal levels related to cycles of ice formation and melting (Manasypov et al., 2015). Further, studies using mass-balance calculations and paleoecological techniques have linked thermokarst erosion in peatlands and the release of Hg into lake surface waters (sub-Arctic

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Sweden, Klaminder et al., 2008; Rydberg et al., 2010). Stable isotope analysis also suggests that the weak recovery of Pb contamination in two sub-Arctic lakes (despite dramatically reduced atmosphere inputs) may be linked to the subsidence of thawing permafrost soils (sub-Arctic Sweden, Klaminder et al., 2010). Permafrost degradation may affect the mobility of inorganic contaminants differently across different Arctic regions. Deison et al. (2012) found that Mackenzie delta upland lakes impacted by the development of retrogressive thaw slumps in siliciclastic soils had lower levels of Hg in surface sediments when compared to reference lakes. In this instance, thaw slumping may have led to a dilution of organic material and associated mercury (Hg) due to high inorganic sedimentation rates. On the other hand, MacMillan et al. (2015) showed that small thermokarst lakes located in sub-Arctic and High Arctic Canada dominated by slumping of organic soils showed elevated concentrations of Hg and toxic methylmercury. This was strongly related to inputs of organic matter and nutrients into surface waters.

In summary, permafrost has historically been considered an impermeable barrier to the movement of contaminants due to their low infiltration and mobility in frozen soils (Grannas et al., 2013). However, ongoing and future climate warming will likely disrupt the sequestration of contaminants in permafrost soils, which leads to enhanced leaching and mobility of organic and inorganic contaminants to nearby aquatic systems. This, however, will not necessarily result in increased contaminant concentrations due to a dilution effect by other materials transported along with the contaminant.

2.2.7. Overarching considerations

Overall, the manner in which permafrost thaw affects surface water chemistry as a result of changing land-to-water fluxes will be dependent on the constituent and the landscape. While evidence suggests that constituents such as suspended sediments and weathering ions will experience neutral to increasing effects in response to permafrost thaw, the effect on constituents such as DOC and nutrients is likely to be more variable. As a result, we must consider local conditions, including soil composition (mineral vs. organic-rich), the nature of thaw (e.g., thermokarst processes on hillslopes vs. within lowlands, vs. active layer deepening), and the current extent of permafrost (continuous vs. discontinuous) to best understand the effects of thaw on land-water fluxes of chemical constituents.

Particularly for streams, hydrological connectivity between sites of thaw and the stream system is also an important consideration, because lateral inputs are typically more influenced by conditions in the riparian and deeper subsurface zones than by conditions at ridge tops and the near surface (Stieglitz et al., 2003; Rastetter et al., 2004). Thus, in areas where permafrost is thawing rapidly, the proximity of thaw features to aquatic systems, and the hydrological conditions of local stream and subsurface zones are likely to have a strong influence on the biogeochemical imprint of lateral inputs to streams. While TEFs have the capacity to move large quantities of soils and nutrients downslope, hydrological connectivity must be present to enable these constituents to reach the stream for biogeochemical impact to occur.

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3 Pathways of degradation

The carbon, nutrients, and contaminants that are delivered to aquatic ecosystems as a result of permafrost thaw will have a critical effect on the functioning of these systems. Changes in biological function (see section 4) will determine the relative balance between processing within freshwater systems, versus loss via potential outflow pathways and the eventual delivery of these constituents to sediments (see section 4.2) or 'downstream' to the Arctic Ocean (see section 4.5). In the case of carbon, differences in quality will affect whether processing and uptake results in the release of greenhouse gases (GHG), or C incorporation into microbially-based food webs. At the same time, changes in stratification, redox, solubility, and oxygen availability (sections 2.1, 2.2, and 3.5.1) will affect the balance between CO₂ and CH₄ release. In this section, we review the dominant pathways of carbon and contaminant degradation in permafrost-thaw impacted systems. We focus specifically on bio- and photo-degradation pathways, and the effect of permafrost thaw on GHG emissions from thaw-impacted systems. The manner in which permafrost thaw affects nutrient uptake within aquatic systems is addressed in section 4.3.

3.1 Biodegradation of organic carbon

In the Arctic, where transfers of organic C from soils to aquatic ecosystems can be especially strong (Kling et al., 1991), C fluxes from surface waters to the atmosphere, and from land to ocean, may represent up to 40% of the net land-atmosphere C exchange (maximum flux of ~0.16 Pg C/y compared to a net terrestrial sink of 0.4 ± 0.4 Pg C/y; McGuire et al., 2009). In many cases, these C fluxes from freshwaters to the atmosphere or to the coastal oceans are supported by the degradation of terrestrially-derived DOC (e.g., Yukon River, Alaska, Spencer et al., 2008), although DIC derived from weathering can also be an important CO₂ source (Yukon River, Alaska, Striegl et al., 2012). Biological processing of DOC occurs prior to, and upon, hydrologic delivery to surface waters (Alaska, Michaelson et al., 1998; NE Siberia, Spencer et al., 2015). The biodegradability of DOC in various Arctic systems is dependent on several factors including DOC source and chemical character (Michaelson et al., 1998; Wickland et al., 2007; 2012; Balcarczyk et al., 2009; Mann et al., 2012; Olefeldt et al., 2013; Abbott et al., 2014), nutrient availability (Holmes et al., 2008; Mann et al., 2012; Wickland et al., 2012; Abbott et al., 2014), water temperature (Wickland et al., 2012), and prior processing (Michaelson et al., 1998; Wickland et al., 2007; Spencer et al., 2015). There are strong seasonal patterns in DOC biodegradability in large Arctic rivers, where the relative amount of biodegradable DOC (BDOC) is greatest in winter and spring, and generally declines through the summer and fall (Holmes et al., 2008; Mann et al., 2012; Wickland et al., 2012; Vonk et al., 2015), reflecting the influences of seasonal thaw depth on DOC sources and hydrologic connectivity. Soil BDOC does not show a strong seasonality (Wickland et al., 2007; Vonk et al., 2015), supporting the notion that changes in DOC residence time and processing in soils prior to delivery to surface waters is a primary control on aquatic BDOC (Striegl et al., 2005; Vonk et al., 2015).

Permafrost presence and extent has direct and indirect influences on biodegradable DOC in aquatic ecosystems through its controls on potential sources and on hydrologic pathways and rate of delivery. A synthesis study by Vonk et al. (2015) of BDOC in circum-Arctic soils and surface waters finds higher BDOC in soils and aquatic systems with increasing permafrost extent. In the absence of

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779 direct slumping, sources of DOC in permafrost-impacted areas are restricted to surface litter and
 780 modern active layer soils, which have relatively high total and labile C contents that are readily
 781 accessible (e.g., Holmes et al., 2008), and thus are strong potential sources of BDOC. Deeper soils
 782 having generally lower C content are more accessible in discontinuous permafrost areas, and
 783 hydrologic flow paths with longer residence times allow for greater opportunity of DOC processing
 784 during transport (Alaska, Walvoord and Striegl, 2007) generally resulting in lower potential BDOC
 785 delivery to aquatic systems (Vonk et al., 2015). However, there are observations of increasing
 786 delivery of biodegradable DOC to aquatic ecosystems in areas of decreasing permafrost extent
 787 (western Siberia, Kawahigashi et al., 2004; Alaska, Balcarczyk et al., 2009), with one explanation
 788 being the preferential sorption of more recalcitrant hydrophobic DOC constituents to increasingly-
 789 exposed/accessible mineral soils (Kawahigashi et al., 2004). Therefore broad generalizations of
 790 permafrost control on BDOC are still difficult to make with certainty.

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791 Permafrost thaw and thermokarst formation can potentially impact biodegradable DOC in aquatic
 792 ecosystems by altering sources, rates, and pathways of hydrologic delivery. Newly thawed soils are
 793 potential DOC sources that can exceed DOC released from seasonally thawed active layer soils (sub-
 794 Arctic Sweden, Roehm et al., 2009; Alaska, Waldrop et al., 2010). Studies of distinct DOC sources
 795 suggest that non-permafrost-derived soil pore water DOC is moderately biodegradable (Wickland
 796 et al., 2007; Roehm et al., 2009; Vonk et al., 2015), whereas certain permafrost soil-derived DOC can
 797 be highly biodegradable, particularly Pleistocene yedoma DOC in NE Siberia and Alaska (Vonk et al.,
 798 2013; Abbott et al., 2014; Spencer et al., 2015). These studies point to a high susceptibility of
 799 permafrost DOC to degradation during transport from soils, and within surface waters, compared to
 800 non-permafrost DOC, with aliphatic DOC originating in permafrost being preferentially degraded
 801 (NE Siberia, Spencer et al., 2015). Surface waters contain a mixture of DOC from different sources,
 802 and therefore it is difficult to isolate the relative biodegradability of permafrost soil vs. non-
 803 permafrost soil (active layer) derived DOC within surface waters (Holmes et al., 2008; Balcarczyk et
 804 al., 2009; Mann et al., 2012; Wickland et al., 2012). Frey et al. (2015) have shown a relatively
 805 constant proportion of bioavailable DOC (~4.4%; based on five day biological oxygen demand
 806 assays) along the flow-path continuum throughout the Kolyma River basin in Siberia. Actively
 807 thawing permafrost features, however, release elevated amounts of BDOC to low order water tracks
 808 and outflows (upt to 40% BDOC after 30-40 days incubation; NE Siberia, Vonk et al., 2013; Alaska,
 809 Abbott et al., 2014), but significant biodegradation during transport to higher order streams and
 810 rivers remove substantial amounts of permafrost DOC before reaching major Arctic rivers and the
 811 ocean (NE Siberia, Mann et al., 2015; Spencer et al., 2015). Increasing hydrologic flow path lengths
 812 and residence time in soils in some areas as a result of permafrost thaw likely promote DOC
 813 processing and sorption within watersheds prior to discharge to surface waters (Yukon River,
 814 Alaska, Striegl et al., 2005), further reducing the likelihood of biodegradable permafrost DOC
 815 reaching aquatic systems.

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816 Thermokarst can also release significant quantities of POC to aquatic ecosystems (section 2.2),
 817 often substantially outweighing the amount of released DOC (assessed via direct measurements, or
 818 assuming a 1-2% conversion between suspended sediments and POC; Bowden et al., 2008; Lewis et
 819 al., 2012; Vonk et al., 2013; Abbott et al., 2015). In streams impacted by thermokarst, POC will
 820 remain entrained for long distances downstream (NW Canada, Kokelj et al., 2013), and may

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834 reasonably be subject to significant biological decomposition as a result of this entrainment, as has
 835 been shown for other, temperate, systems (Richardson et al., 2013). Degradation of the POC
 836 delivered to aquatic systems as a result of permafrost thaw, however, has received little attention to
 837 date.

838 3.2 Photodegradation of organic carbon

839 Recent studies indicate that sunlight can play an important role in dissolved organic matter (DOM)
 840 degradation in thermokarst-impacted lakes and ponds (Alaska, Cory et al., 2013, 2014; Canadian
 841 High Arctic, Laurion and Mladenov, 2013; sub-Arctic Sweden, Koehler et al., 2014). To understand
 842 why sunlight can be an important control on DOM degradation in these systems, we review controls
 843 on DOM photo-degradation and the specific characteristics of thermokarst lakes that can maximize
 844 opportunities for photo-degradation.

845 Sunlight breaks down DOM into three broad classes of products: (1) CO₂ (and CO that can be
 846 subsequently oxidized to CO₂; photo-mineralization), (2) partially oxidized or degraded DOM that
 847 bacteria can then readily respire to CO₂ (photo-stimulated bacterial respiration), and (3) partially
 848 oxidized or degraded DOM that is more recalcitrant to bacteria (Alaska, Cory et al., 2010, 2013,
 849 2014; Hong et al., 2014; Canadian High Arctic, Laurion and Mladenov, 2013). Where the presence of
 850 thermokarst increases the transfer of old permafrost carbon from land to water, photochemical
 851 conversion of DOC to CO₂ and photo-stimulated bacterial respiration may therefore increase old C
 852 transfer to the atmosphere. This occurs on relatively short time scales (e.g. months to decades),
 853 thus providing a positive feedback to global warming (e.g., Cory et al., 2013, 2014; Laurion and
 854 Mladenov, 2013). The water column rate of DOM photo-degradation to CO₂ or to partially oxidized
 855 DOM increases with increasing UV radiation, and also depends on the rate of light absorption by
 856 DOM in the water column, and the lability of DOM to be converted to CO₂ or to partially oxidized
 857 DOM during light exposure.

858 UV irradiance entering the water column depends on the extent and presence of snow and ice cover
 859 (Vincent and Belzile, 2003), the solar zenith angle (i.e., latitude, date and time of day), and the
 860 composition of the atmosphere (e.g., ozone, and the amount and type of clouds and aerosols)
 861 (Vavrus et al., 2010; Bernhard et al., 2013). In surface waters across the Alaskan Arctic (from Toolik
 862 Lake to Barrow, AK), the sun is above the horizon from approximately mid-May through mid-July,
 863 but ~ 90% of the daily UV flux involved in DOM degradation reaches surface waters during the day
 864 due to the low solar zenith angle overnight (Alaska, Cory et al., 2014). Clouds generally decrease
 865 surface UV, but the effect of clouds can be offset by ozone levels, making it difficult to predict
 866 surface UV based only on latitude and date across the Arctic (Vavrus et al., 2010; Bernhard et al.,
 867 2013). Although less UV generally reaches surface waters in the Arctic compared to lower latitudes
 868 due to lower solar zenith angles, thermokarst lakes and ponds can often have high concentrations
 869 of light-absorbing DOM that is susceptible to photo-degradation, thus potentially counter-balancing
 870 the lower UV (Alaska, Cory et al., 2014; sub-Arctic Sweden, Koehler et al., 2014).

871 The many shallow ponds and lakes across the Arctic often contain high concentrations of light-
 872 absorbing CDOM (e.g., Alaska, Hobbie, 1980; NW Canada, Gareis et al. 2010; sub-Arctic Québec,
 873 Watanabe et al., 2011; Alaska, Cory et al., 2014) as is consistent with high concentrations of light-

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887 [absorbing DOC draining from soils surrounding these lakes \(Alaska, Judd et al., 2007; Merck et al.,](#)
 888 [2012; NE Siberia, Frey et al., 2015\)](#). For example, absorption coefficients for CDOM were reported
 889 to range from 9 - 17 m⁻¹ at 330 nm in lakes of the Mackenzie Delta (Gareis et al., 2010), while a
 890 survey of 380 lakes in the Alaskan Arctic reported a mean of 11 m⁻¹ vs. 31 m⁻¹ at 320 nm for lakes
 891 near the foothills of the Brooks Range vs. lakes on the coastal plain, respectively (Cory et al., 2014).
 892 For small [sub-Arctic](#) thermokarst [lakes at the southern limit of permafrost along the Eastern](#)
 893 [Hudson Bay, Watanabe et al. \(2011\)](#) present a large range of absorption coefficients at 320 nm (9.9
 894 to 56 m⁻¹) at the southern limit of permafrost along eastern Hudson Bay, while even a wider range
 895 was obtained by Breton et al. (2009) [in the same region](#), (reaching up to 171 m⁻¹), although these
 896 systems are typically also affected by high levels of non-CDOM UV absorbance (see below).

897 Near-surface rates of DOM photo-degradation increase linearly with increasing CDOM, while the
 898 depth-integrated rates of photo-degradation in the water column depend non-linearly on CDOM
 899 concentrations due to the attenuation of light (Miller, 1998). In thermokarst lakes [and ponds](#), CDOM
 900 is often the main UV-absorbing constituent (e.g., [Hobbie, 1980](#); Gareis et al., 2010; [Cory et al., 2014](#)),
 901 and thus controls the depth of UV light penetration. However, non-algal particles and especially fine
 902 inorganic particles can contribute a significant portion of UV attenuation in thermokarst [lakes](#)
 903 influenced by marine clays and silts (Watanabe et al., 2011). In such cases, UV is attenuated to a
 904 much larger extent, limiting photochemical reactions to the very surface under the strongly
 905 stratified conditions observed for these systems (Laurion et al., 2010). Although CDOM
 906 concentrations, and thus light attenuation, are often high in lakes [and ponds](#) across the Arctic, the
 907 whole water column can still be exposed to UV because many of these systems are shallow ([Gareis](#)
 908 [et al., 2010](#); Cory et al., 2014; [see also section 2.1](#)). For example, a survey of CDOM and UV light in
 909 thermokarst lakes of the Mackenzie River delta concluded that 19% and 31% of the water column
 910 was exposed to UVB and UVA radiation, respectively (Gareis et al., 2010). For a series of
 911 thermokarst lakes [and ponds](#) of the coastal plain in the Alaskan Arctic, up to 20% of the water
 912 column was exposed to UVB while 30 - 100% was exposed to UVA (Cory et al., 2014). Exceptions
 913 include turbid streams, [ponds](#), and lakes impacted by thermokarst slumping (Bowden et al., 2008;
 914 [Gareis et al., 2010; Watanabe et al., 2011](#); Cory et al., 2013), or [ponds](#) with abundant macrophyte
 915 production (Gareis et al., 2010), where UV penetration is low.

916 The degree to which DOM drained from catchment soils underlain by permafrost is susceptible to
 917 photo-degradation, quantified as the apparent quantum yield for each major class of DOM photo-
 918 products, has been measured to be on the high end of the range reported for aquatic DOM ([Alaska,](#)
 919 Cory et al., 2013, 2014; Hong et al., 2014; [sub-Arctic Sweden](#), Koehler et al., 2014). These findings
 920 suggest that DOM originating from soils underlain by permafrost may be more labile to photo-
 921 degradation relative to DOM in freshwaters outside the Arctic, consistent with prior studies
 922 [showing a high photo-reactivity of DOM in Arctic surface waters \(NE Siberia, Mann et al., 2012;](#)
 923 [Canadian High Arctic, Laurion and Mladenov, 2013\)](#) [that is suggested to increase downstream in](#)
 924 [the network \(Frey et al., 2015\)](#). However, DOM leached specifically from the permafrost soil layer
 925 sampled across the Arctic has a consistently lower concentration of aromatic, light absorbing
 926 carbon (i.e., lower CDOM per DOC concentration), often quantified as lower SUVA₂₅₄ values ([Mann](#)
 927 [et al., 2012; Cory et al., 2013, 2014; Abbott et al., 2014; Ward and Cory, 2015](#)), compared to DOM
 928 draining from the active, organic surface layer. Despite lower concentrations of light-absorbing C,

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952 permafrost DOM has been measured to be equally or more sensitive to photo-degradation in a
 953 series of sites in the Alaskan Arctic (i.e., when comparing apparent quantum yields for
 954 photochemical CO₂ production for example, corrected for differences in rates of light-absorption;
 955 Cory et al., 2013). These results may suggest that the chemical composition of permafrost DOM
 956 makes it more reactive to sunlight than expected based on aromatic C content alone.

957 Overall, the typical range of CDOM concentrations and the shallow water depths in thermokarst
 958 lakes and ponds can mean that a greater fraction of DOM is exposed to UV in the water column
 959 compared to deeper, non-thermokarst lakes. Especially for lakes and ponds with no outlet, the
 960 residence time of DOM and its exposure to UV light is high, thus confining DOM to a thin boundary
 961 layer where opportunities for photo-degradation are maximized (e.g., Alaska, Loughheed et al., 2011;
 962 sub-Arctic Sweden, Olefeldt and Roulet, 2012). The alternation of stratification periods (intensive
 963 UV exposure at the very surface) with night-time cooling and mixing (renewal of surface water
 964 DOM) observed in many shallow thermokarst lakes may also offer a greater opportunity for
 965 efficient DOM photo-degradation. With forthcoming climate change and deeper permafrost thaw, C
 966 flux from peaty soils to thermokarst lakes may also be enhanced in some regions, and the released
 967 DOM will be subject to UV-induced mineralization, especially as the summer season lengthens
 968 (Erickson et al., 2015). On the other hand, a study of 73 lakes in the Mackenzie Delta uplands found
 969 that slump-impacted lakes had significantly lower CDOM than unimpacted lakes (Kokelj et al., 2009;
 970 Thompson et al., 2012; see section 2.2), indicating that photodegradation may decline in some
 971 slump-impacted systems due to adsorption of CDOM to basic cations and clay particles. A strong
 972 response in UVB attenuation to small changes in CDOM was observed in lakes of NW Finnish
 973 Lapland, suggesting that even minor shifts in CDOM may largely change the UV radiation exposure
 974 of high-latitude lakes, with likely consequences on the photochemistry and biota (Forsström et al.,
 975 2015). To understand the role of sunlight in DOM processing in thermokarst waters, future work
 976 must quantify UV irradiance in the water column, residence time of DOM in the UV-exposed portion
 977 of the water, and identify the factors that control vertical losses of DOM and the lability of
 978 permafrost DOM to photo-degradation.

979 3.3 Photochemical and microbial transformation of contaminants

980 There are likely to be multiple effects of permafrost thaw on the mobility and transformation of
 981 contaminants in Arctic environments. The climate-triggered release of contaminants may increase
 982 contaminant transport into aquatic systems through increased leaching and hydrological
 983 connectivity. Thermokarst lakes with anoxic sediments may cause the remobilization of
 984 contaminants, and may also allow enhanced microbial activity and hypolimnia, and bacterial metal
 985 alkylation in the hypolimnion (e.g., production of the neurotoxin methylmercury). Here, we
 986 examine the potential mechanisms that may allow Arctic warming to lead to increases in the
 987 degradation and transformation of contaminants.

988 Photochemical transformations may affect both the mobility and availability of photoreactive
 989 contaminants such as the DOC-driven photoredox transformations of As (Buschmann et al., 2005)
 990 and Hg. Tseng et al. (2004) showed that Hg can be photoreduced and volatilized in Alaskan surface
 991 waters, whereas the neurotoxin methylmercury in the same region can be photodegraded

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1001 (Hammerschmidt and Fitzgerald, 2010). Photochemical transformation of contaminants may
1002 become more important in regions where the total area of non-turbid thermokarst ponds is
1003 increasing, as these shallow ecosystems are irradiated constantly during the polar summer (e.g.,
1004 Mann et al., 2014).

1005 Changes in the release of DOC and POC into aquatic ecosystems may significantly affect the phase
1006 partitioning and solubility of contaminants. In discontinuous permafrost regions, an increase in
1007 labile organic carbon (i.e. DOC) with thawing may lead to enhanced microbial activity and hence
1008 alter the microbial transformation of some contaminants (Roehm et al., 2009). Microbes can
1009 influence contaminant cycling by degrading organic contaminants, alkylating metals, and creating
1010 redox gradients that may modify the mobility and toxicity of toxic metals. Permafrost thaw can
1011 affect microbial diversity and microbial activity. Microbial diversity typically is highest in the
1012 surface active layer and decreases towards the permafrost table (NW Canadian Arctic, Frank-Fahle
1013 et al., 2014). hence deepening of the active layer will likely modify microbial diversity. Changing
1014 microbial diversity in combination with nutrient and temperature effects can affect the microbial
1015 degradation of organic pollutants in thawing permafrost (Bell et al., 2013). Microbial activity can be
1016 enhanced when permafrost thaw creates new environments such as warm, stratified thermokarst
1017 lakes, which may be potential sites for bacterial metal alkylation (Stern et al., 2012). such as found
1018 in two regions of NW Canada (MacMillan et al., 2015). Thermokarst lakes with hypoxic or anoxic
1019 bottom waters may be sites that are highly conducive to microbial Hg(II) methylation, and
1020 increasing inputs of organic matter and nutrients from thawing permafrost into these systems may
1021 have potentially important consequences for the transport or in situ production of methylmercury
1022 (MacMillan et al., 2015).

1023 Manasypov et al. (2014) showed that thermokarst lakes in Siberia have close relationships between
1024 diagenetic processes and the remobilization of contaminants from the sediments. This
1025 remobilization is tied to diagenetic reactions occurring in these lakes due to the microbial
1026 mineralization of natural organic matter. The anoxic conditions in lake sediments during the early
1027 stages of thermokarst lake development result in microbe-mediated reactions causing authigenic
1028 sulphide precipitation (i.e., the reduction of sulphate) that can create a sink for metals in the
1029 sediments (western Siberia, Audry et al., 2011). Early diagenetic reactions in Siberian thermokarst
1030 lakes and the resulting shift in redox conditions are responsible for the partitioning of trace
1031 elements, including several major contaminants (As, Cu, Zn, Cd, Pb, Ni). During all stages of lake
1032 development in this region, the sediments may be a source of dissolved Ni and As to the water
1033 column (Audry et al., 2011).

1034 As outlined in section 2.2.6, permafrost thaw may initially lead to a higher mobility of organic and
1035 inorganic contaminants into aquatic systems across different Arctic regions. For instance, peat
1036 leaching leads to higher levels of dissolved trace metal concentrations in Siberian thermokarst
1037 lakes (Manasypov et al., 2014). However, the bioavailable metal pool will ultimately be controlled
1038 not only by the importance of the dissolved fraction, but also by chemical and biological changes,
1039 such as photodegradation and organic matter complexation. For inorganic contaminants, the
1040 quantity, quality, and molecular weight of organic matter being released, photolyzed, and

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1058 [microbially transformed from thawing permafrost during thermokarst lake evolution is likely a key](#)
1059 [driver of metal bioavailability in these systems](#) (Pokrovsky et al., 2011).

1060 **3.4 Microbiology of thaw waters**

1061 Biogeochemical data collected to date from thermokarst lakes, rivers, [ponds](#), and wetlands point to
1062 the importance of these environments as sites of intense microbial activity in the northern
1063 landscape. As a result there is an increasing effort to apply molecular microbiological methods,
1064 particularly next generation nucleic acid sequencing techniques, to understand the biodiversity,
1065 network relationships, and biogeochemical capabilities of these microbial communities. This
1066 research theme is still at an early stage of development, but the picture that is emerging is one of
1067 complex microbial consortia, with all domains of life well represented, and dominance by certain
1068 groups that play key biogeochemical roles (e.g., [Negandhi et al., 2013](#); [Crevecoeur et al. 2015](#);
1069 [Przytulska et al., 2015](#)).

1070 Methanogenic archaea occur in high abundance in the anoxic, CH₄-rich waters of permafrost thaw
1071 waters and wetlands, and molecular techniques have revealed a variety of taxa. In the [Canadian](#)
1072 [High Arctic](#), gene signatures from acetoclastic and hydrogenotrophic methanogenic Archaea were
1073 detected in ponds associated with ice-wedge polygons ([Negandhi et al., 2013](#)), while [Mondav et al.](#)
1074 [\(2014\)](#) showed across a permafrost gradient in northern Sweden that partially thawed sites were
1075 often dominated by a single taxon (*Methanoflorens stordalenmirensis*) that belongs to the
1076 uncultivated archaeal lineage 'Rice Cluster II'. Metagenomic analysis showed that this micro-
1077 organism has the genes for hydrogenotrophic methanogenesis. A subsequent molecular study by
1078 [McCalley et al. \(2014\)](#) in sub-Arctic Sweden combined with isotopic analyses showed that the
1079 abundance of this taxon is a predictor of the relative proportions of carbon released from the
1080 thawing permafrost as CH₄ versus carbon CO₂.

1081 There is now a rapidly growing DNA data base for bacteria in permafrost soils, which often contain
1082 anaerobic groups such as sulphate reducers, Fe(III) reducers, and denitrifiers, and many aerobic
1083 groups including actinobacteria and methanotrophs ([Jansson and Tas, 2014](#)). By comparison, much
1084 less is known about the microbial constituents of thaw waters. Soil crusts in the [High Arctic](#) polar
1085 desert have been shown to contain remarkably diverse communities of bacteria, with evidence that
1086 their populations of cyanobacteria and acidobacteria are stimulated by water track flows over the
1087 permafrost ([Steven et al., 2013](#)). High throughput analysis of bacterial samples from [High Arctic](#)
1088 [ponds](#) showed that the planktonic sequences in these waters were dominated by carbon degrading
1089 taxa in the Bacteroidetes, Betaproteobacteria and Actinobacteria ([Negandhi et al., 2014](#)). In
1090 contrast, the sediment community had a higher alpha-diversity and the sequences included carbon
1091 degraders (29–46 %), cyanobacteria (20–27 %), purple non-sulfur bacteria (6–13 %),
1092 methanotrophs (11–20 %), and methanogen symbionts (1–2 %).

1093 DNA clone library analysis of thermokarst lakes in a sporadic permafrost region in sub-Arctic
1094 [Québec](#) revealed large differences in the assemblages inhabiting the different water layers, and the
1095 presence of methanotrophic bacteria ([Rossi et al., 2013](#)). Subsequent analysis of lake communities
1096 in the same region by high throughput RNA sequencing showed that the dominant bacterial taxa
1097 were beta-proteobacteria, especially the genera *Variovorax* and *Polynucleobacter* (both known to

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1119 degrade a wide variety of organic compounds), and that methanotrophs (notably *Methylobacter*)
 1120 were also well represented (Crevecoeur et al., 2015). Methanotrophic taxa accounted for up to 27%
 1121 of the total bacterial sequences, indicating the importance of CH₄ as an energy source in these
 1122 ecosystems. A puzzling observation was that the anoxic bottom waters in most of these
 1123 thermokarst Jakes had abundant methanotrophs, accounting for up to 23% of the sequences. This
 1124 could be the result of intermittent injection of oxygen into these bottom waters by mixing, or
 1125 sustained viability of the methanotrophs mixed down from the aerobic surface zone. Such mixing
 1126 occurs mostly during fall in these sub-Arctic lakes (Deshpande et al., 2015), which would imply
 1127 prolonged survival under anoxic conditions, and the availability of an inoculum for rapid response
 1128 to oxygen resupply during mixing. However, few data are available from thermokarst lakes further
 1129 north, and it is not known whether these sub-Arctic patterns occur elsewhere. A major unknown for
 1130 thermokarst lakes throughout the sub-Arctic and Arctic is the composition of winter microbial
 1131 communities beneath the ice, and this will require close attention in the future.

1132 High-throughput DNA sequencing has also been used to examine biogeographical patterns. The
 1133 bacterial composition of thermokarst Jakes was examined over a North-South gradient of
 1134 permafrost degradation in sub-Arctic Québec, showed that greater differences occurred among
 1135 valleys across this gradient than among Jakes within a valley, despite marked differences in
 1136 limnological properties among neighbouring Jakes (Comte et al., 2015). This implies that the
 1137 taxonomic composition and perhaps also the biogeochemical functioning of thermokarst Jake
 1138 bacterial assemblages are regulated by local landscape features, such as the extent of permafrost
 1139 thaw.

1140 Phototrophic organisms in thaw waters include photosynthetic sulphur bacteria (sub-Arctic
 1141 Québec, Rossi et al., 2013; Crevecoeur et al., 2015), benthic cyanobacteria, purple non-sulphur
 1142 bacteria (Canadian High Arctic, Negandhi et al., 2014), and picocyanobacteria, as well as eukaryotic
 1143 algae of diverse phylogenetic groups (sub-Arctic Québec, Przytulska et al., 2015), however the
 1144 network associations among these organisms and other microbial taxa has yet to be explored. Such
 1145 analyses have been applied to temperate lake communities, combining bacterial DNA sequence data
 1146 with phytoplankton and zooplankton counts by microscopy, and these reveal highly connected,
 1147 potential keystone taxa in the mixed communities (Peura et al., 2015).

1148 Large Arctic river systems receive thaw waters from throughout their catchments and can serve to
 1149 monitor large-scale patterns. Crump et al. (2009) reported that bacterial communities showed a
 1150 large spatial synchrony, along with clear seasonal community differences driven by shifts in
 1151 hydrology and biogeochemistry that reassembled annually. Furthermore, Crump et al. (2012)
 1152 observed a decreasing species diversity downslope in a soil-stream-lake sequence in Alaska. Soil
 1153 waters and headwater streams showed highest species richness, whereas lake waters show a lower
 1154 diversity. They suggest that bacterial and archaeal diversity in freshwaters is initially structured by
 1155 inoculation of soil microbes, and then subject to a species-sorting process during downslope
 1156 dispersal. Permafrost thaw could lead to a greater transfer of soil microbes into aquatic
 1157 communities.

1158 A conspicuous gap in information available to date is the diversity and role of viruses in these
 1159 microbe-rich habitats. Viruses are likely to be the biologically most abundant particles in these

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Deleted: However, the composition of winter microbial communities in ice-covered thermokarst lakes is at present unknown, and will require close attention in the future.

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waters, as elsewhere, and may influence the species succession of microbes in all domains of life, affect carbon cycling by their lytic activities, and have a controlling influence on evolutionary processes via horizontal gene transfer (Suttle et al., 2007). New viral lineages are being discovered in environments elsewhere (e.g., freshwaters in the Canadian High Arctic, Chénard et al., 2015), and thermokarst lakes and ponds will likely yield additional new groups given the diversity of potential host taxa.

3.5 Aquatic gas fluxes

3.5.1 Emission of CO₂ and CH₄ from permafrost-thaw impacted systems

In well-drained terrestrial environments, permafrost thaw leads to microbial decomposition resulting in variable production and emission of CO₂ (e.g., Schuur et al., 2009; Schädel et al., 2014). Thaw of ice-rich permafrost, particularly in poorly drained lowland areas, results in ground subsidence and saturated soils that take the form of thermokarst lakes, wetlands, and slumping into streams. Since waterlogging slows the diffusion of oxygen from the atmosphere into soils, this results in anoxic conditions in sediments as well as in portions of the overlying water columns of many thermokarst water bodies. Under anaerobic conditions, decomposition of organic matter also produces CH₄ (e.g., Alaska, Wickland et al., 2006). Where soils surrounding thermokarst lakes are anoxic, lateral inputs of CH₄ produced within the active layer can also occur (Alaska, Paytan et al., 2015). Work in northern Siberia suggests that small water bodies may be particularly important for CO₂ and CH₄ emissions (Repo et al. 2007; Abnizova et al. 2012). In a study on northern Ellesmere Island, Canada, desert soils consumed CH₄ during the growing season, whereas the wetland margin emitted CH₄, with an overall positive CH₄ flux over the landscape using, varying with soil temperature (Emmerton et al., 2014). The CH₄ flux varied closely with stream discharge entering the wetland and hence the extent of soil saturation.

In a study of 40 Alaskan thermokarst lakes (Fig. 5) that span large gradients of climate, vegetation, geology, and permafrost regimes, Sepulveda-Jáuregui et al. (2015) found that all lakes were net sources of atmospheric CH₄ and CO₂ (when integrated over a year) as also noted earlier by Kling et al. (1991; 1992). On a C mass basis, CO₂ emissions from Alaskan lakes were ~6-fold higher than CH₄ emissions. However, considering the ~30-fold stronger global warming potential of CH₄ vs. CO₂ over 100 years (GWP₁₀₀; Myhre et al., 2013), CH₄ emissions had nearly twice the impact on climate as CO₂ emissions in this region.

In the Eastern Canadian Arctic, a thermokarst lake (deep enough to have unfrozen water in winter and likely a talik underneath) was shown to be a relatively small GHG emitter in July (Bouchard et al., 2015), although its thermal structure suggests that GHG potentially stored in the hypolimnion is transferred to the atmosphere at the autumnal overturn. Large variations in summertime CO₂ and CH₄ fluxes were shown in smaller lakes and ponds from two sites located in the Canadian sub- and High Arctic (Laurion et al., 2010). Turbid, sub-Arctic thermokarst lakes were all GHG emitters, but showed on average a 530-fold higher CO₂ than CH₄ diffusive flux in summer, with strong GHG gradients in the hypolimnion (summer storage). In the High Arctic, polygonal ponds over low-centered ice wedge polygons were CO₂ sinks because of colonization by active cyanobacterial mats (Laurion et al., 2010), while shallower ice-wedge trough ponds were identified as the main GHG

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emitters (Negandhi et al., 2013), with summer CO₂ fluxes ~25-fold higher than CH₄ diffusive flux. At this site, the CH₄ ebullition flux (likely background ebullition) was in the same range as the diffusive flux (Bouchard et al., 2015).

In streams and rivers, emission of CO₂ is typically much greater than emission as CH₄ (Yukon River, Alaska, Striegl et al., 2012). On a catchment scale, sub-Arctic and Arctic streams within permafrost zones can emit relatively high amounts of GHG relative to their areal extent (sub-Arctic Québec, Teodoru et al., 2009; Alaska, Striegl et al., 2012; Siberia, Crawford et al., 2013; Denfeld et al., 2013; sub-Arctic Sweden, Lundin et al., 2013), and gaseous emissions can account for up to 50% of total C exports (Striegl et al., 2012). For example, in northern Sweden streams accounted for 4% of the aquatic surface area yet accounted for 95% of the total aquatic emissions (Lundin et al., 2013), whereas in northern Québec streams accounted for 1% of the aquatic surface and accounted for 25% of the aquatic emissions (Teodoru et al., 2009). Stream CH₄ emissions can also be significant to total catchment emissions; for example in interior Alaska stream CH₄ emission was estimated to be up to 10% of catchment terrestrial emissions despite the very low surface area (<0.2% of catchment area; Crawford et al., 2013). The relatively high emissions can be attributed both to supersaturation relative to the atmosphere as well as high gas transfer velocities associated with these more turbulent waters (Kling et al., 1992; Striegl et al., 2012; Denfeld et al., 2013; Lundin et al., 2013). To date, however, there are no published studies to show how gas fluxes are affected by the direct action of thermokarst slumping into streams.

A global-scale database of 4902 lakes have previously shown a significant relationship between DOC and CO₂ (Sobek et al., 2005). This relationship has also been shown for a smaller set of Arctic and sub-Arctic thermokarst lakes in Canada, especially for the chromophoric fraction of DOM (Laurion et al., 2010). Sepulveda-Jáuregui et al. (2015) also found a significant relationship between CH₄ diffusive flux and phosphorus concentrations in a series of thermokarst lakes over a N-S Alaska transect. Therefore, we can assume that when thermokarst slumping leads to an associated increase in DOM and nutrients, we can expect an overall rise in GHG emissions from aquatic systems.

3.5.2 Scale and distribution of GHG measurements

Since the solubility of CO₂ exceeds that of CH₄, CO₂ evades aquatic ecosystems primarily by diffusion, while CH₄ more readily comes out of solution, forming bubbles in sediments that escape to the atmosphere by ebullition. Emission of CH₄ through diffusion from aquatic systems can, however, also be high, particularly in wetlands, lakes, and other standing open water (Alaska, Reeburgh et al., 1998; sub-Arctic Sweden, Lundin et al., 2013). Due to large heterogeneity in the spatial and temporal dynamics of ebullition, this mode of CH₄ emission is less commonly studied than diffusion (Bastviken et al., 2011; Wik et al., in review), although ebullition has been found to be the dominant form of CH₄ emission in many thermokarst lakes (Bartlett et al., 1992; Walter et al., 2006; Sepulveda-Jáuregui et al., 2015; see also discussion on the Eastern Canadian Arctic, above). Recent studies focusing on ebullition dynamics in thermokarst lakes distinguished multiple sub-modes of ebullition emission including seep ebullition, background ebullition, and ice-bubble storage (Alaska and Siberia, Walter et al., 2006; Greene et al., 2014; Langer et al., 2015; Sepulveda-Jáuregui et al., 2015; Fig. 5). Background ebullition is most commonly reported in the literature and

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1288 consists predominately of distributed bubbling from seasonally warm surface sediments. In
 1289 contrast, seep ebullition involves bubbling of CH₄ formed at depth in dense sediments (Fig. 4),
 1290 which are typically found in thaw bulbs beneath thermokarst lakes and streams (Walter Anthony et
 1291 al., 2014). Seep ebullition occurs repeatedly from the same point-source locations and occurs year-
 1292 round due to the thermal lag that results in warmer temperatures in deep sediments through the
 1293 fall to winter. Bubbling rates in hotspot seeps are high enough to maintain open holes in
 1294 thermokarst lake ice, resulting in the emission of CH₄-rich bubbles to the atmosphere throughout
 1295 winter (Zimov et al., 2001; Greene et al., 2014; Fig. 5). In thermokarst lakes where both seep and
 1296 background ebullition were measured, seep ebullition was found to dominate CH₄ emissions
 1297 despite occupying only a small fraction of the lake surface area (Walter et al., 2006). More recently,
 1298 ice-bubble storage, the release of ebullition bubbles seasonally trapped by winter lake ice upon
 1299 spring melt, was also recognized as an important, additional mode of ebullition. It contributed 9-
 1300 13% of total annual CH₄ emissions from thermokarst (and non-thermokarst) lakes in Alaska
 1301 (Greene et al., 2014; Sepulveda-Jáuregui et al., 2015) and was also recognized as an important
 1302 springtime emission mode in West Siberian lakes (Golubyatnikov and Kazanste, 2013). Ice-bubble
 1303 storage is likely an important mode of emission in many Arctic systems since CH₄-rich ice-bubbles
 1304 have been observed in aquatic systems in Northeast Siberia (Walter et al., 2006; Langer et al.,
 1305 2015), Sweden (Wik et al., 2011; Boereboom et al., 2012), Finland (Walter Anthony et al.,
 1306 unpublished data); Greenland (Walter Anthony et al., 2012), Alaska (Walter et al., 2007; Brosius et
 1307 al., 2012; Sepulveda-Jáuregui et al., 2015) and Canada (Duguay et al., 2002; Brosius et al., 2012).

1308 One promising technique for measuring aquatic gas fluxes in permafrost-impacted systems is eddy
 1309 covariance (EC), but EC data on inland freshwater ecosystems are still rare. Currently, on-going EC
 1310 measurements focus on CO₂ and CH₄ fluxes over thermokarst lakes in Siberia (T. Sachs, personal
 1311 communication, 2015; L. Beileli-Marchesini, personal communication, 2015) or sub-Arctic lakes
 1312 within thawing permafrost environments (M. Jammet, personal communication, 2015). Using EC,
 1313 Eugster et al., (2003) found efflux rates of 114 mg C/m²/d over an Arctic Alaskan lake in late July,
 1314 which agreed well with two other continuous flux estimation techniques (boundary layer and
 1315 surface renewal models). In a Swedish boreal lake, CO₂ effluxes determined by episodic floating
 1316 chamber measurements were about 100% larger than fluxes measured with EC, suggesting
 1317 potential biases related to inadequate spatial and/or temporal sampling intervals of the chamber
 1318 method (Podgrajsek et al., 2014a).

1319 While proving the feasibility of EC measurements in freshwater ecosystems, aquatic EC work also
 1320 highlights challenges related to the application of this terrestrially-optimized approach to aquatic
 1321 systems (Vesala et al., 2006; Eugster et al., 2011). Overall, EC shows great promise with respect to:
 1322 (1) integration of all gas flux pathways from the lake sediments to the atmosphere, (2) continuous
 1323 flux monitoring over time, enabling the capture of episodic ebullition events of CH₄ in lakes
 1324 (Eugster et al., 2011), and (3) the analysis of dynamic responses of lake-atmosphere carbon fluxes
 1325 to temporal (including diurnal) changes in environmental variables (Eugster, 2003; Vesala et al.,
 1326 2006; Podgrajsek et al., 2014b). A significant portion of gaseous carbon emissions from seasonally
 1327 ice-covered lakes appears to occur during spring ice-thaw (e.g. Karlsson et al., 2013), stressing the
 1328 importance of year-round carbon flux monitoring on thermokarst lakes.

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1343 3.5.3 Lake morphology and evolution

1344 The morphological diversity of thermokarst lakes will have important consequences for hydrology,
1345 physicochemistry, and thus ultimately the microbial processes responsible for GHG production and
1346 evasion dynamics at the air-water interface. Within the context of permafrost soil organic carbon
1347 content, thermokarst lakes have been classified depending on whether they are surrounded by
1348 yedoma-type permafrost or non-yedoma substrates (Walter Anthony et al., 2012; Sepulveda-
1349 Jáuregui et al., 2015). Yedoma is typically thick (tens of meters), Pleistocene-aged loess-dominated
1350 permafrost sediment with high organic carbon (~2% by mass) and ice (50-90% by volume)
1351 contents (Zimov et al., 2006). When yedoma thaws and ground ice melts, deep thermokarst lakes
1352 with high CH₄ production potentials form (Zimov et al., 1997; Kanevskiy et al., 2011; Walter
1353 Anthony and Anthony, 2013). Because these deep (>2m) lakes are often humic and underlain by a
1354 talik, they are stratified for most of the year and are likely to have an anoxic hypolimnion
1355 controlling GHG producers and consumers. These systems present large GHG seepage ebullition
1356 throughout the year, with a characteristic seasonal pattern in GHG evasion (Walter et al., 2006;
1357 Sepulveda-Jáuregui et al., 2015). Smaller but very turbid thermokarst lakes studied in the Eastern
1358 coast of Hudson Bay also do not freeze to the bottom and can similarly be highly stratified and
1359 anoxic (Laurion et al., 2010; Deshpande et al., 2015).

1360 Some non-yedoma permafrost soils can also have high organic carbon and excess ice
1361 concentrations within several meters of the ground surface; however, these organic- and ice-rich
1362 permafrost horizons are typically thinner than yedoma deposits (Ping et al., 2008; Tarnocai et al.,
1363 2009; Bouchard et al., 2015). As a result, thermokarst lakes formed in non-yedoma permafrost soils
1364 are commonly shallower than yedoma lakes and have been shown to emit less CH₄ (West and Plug,
1365 2008; Grosse et al., 2013; Walter Anthony and Anthony, 2013). For instance, CH₄ emissions from
1366 thermokarst lakes formed in carbon-rich yedoma permafrost were 6-fold higher than emissions
1367 from other lake types across Alaska (Sepulveda-Jáuregui et al., 2015).

1368 Shallow thermokarst lakes (e.g., Yukon, Turner et al., 2014) may allow colonization by plants and
1369 benthic photosynthesizing mats, creating CO₂ sink periods while they remain CH₄ emitters
1370 (Mackenzie delta, Tank et al., 2009; Eastern Canadian High Arctic, Laurion et al., 2010; Negandhi et
1371 al., 2014). These lakes can freeze to the bottom (no talik, depending on latitude), which limits active
1372 GHG production to the unfrozen period of the year. In these shallow lakes there is less opportunity
1373 for the dissolution of ebullitive CH₄ before it escapes to the atmosphere, and thus for its
1374 consumption by methanotrophic bacteria. Furthermore, large and shallow lakes are generally
1375 polymictic with GHG evasion largely influenced by winds, generating oxic conditions. For very small
1376 water bodies (a few m²) such as ice-wedge trough ponds, microtopography will be the main
1377 regulator of thermal structure, and gas exchange will be most affected by heat flux. Even though
1378 trough ponds are very shallow systems (<1 m), they can be highly stratified with only occasional
1379 mixing events during the summer (Bouchard et al., 2015), resulting in large periods of hypoxic to
1380 anoxic bottom waters, and evasion of GHG stored in bottom waters following changes in
1381 meteorological conditions. Depending on their erosional features these ponds can also be colonized
1382 by aquatic plants associated with efficient methanotrophic communities (e.g., Siberia, Liebner et al.,
1383 2011).

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1395
1396 Finally, the evolution of thermokarst lake landscapes is a critical determinant of the overall carbon
1397 balance of these systems (van Huissteden et al., 2011; Walter Anthony et al., 2014). This landscape
1398 evolution can be observable at timescales on the order of 30-40 years (Smith et al., 2005; Bryksina
1399 et al., 2011; Polishuk et al., 2012) and is characterized by cyclical flooding of vegetated soils and
1400 recolonization of drained lake bottoms, and an evolution from strong CH₄ emission during the
1401 initial phase of lake formation, through a phase of carbon accumulation associated with higher
1402 within-lake primary production and the creation of terrestrial wetlands as lakes drain (Ovenden,
1403 1986; van Huissteden et al., 2011; Walter Anthony et al., 2014).

1405 4. Consequences

1406 4.1 Release of old permafrost OC into aquatic systems and the atmosphere

1407 4.1.1 Release of old permafrost OC into aquatic systems

1408 A recent study of Eurasian Arctic river basins by Feng et al. (2013) concluded that climate change-
1409 induced mobilization of old permafrost OC is well underway in the Arctic. In this section we review
1410 the evidence that currently exists for the release of old permafrost OC into aquatic systems.

1411 Mobilization of old permafrost OC to surface waters could occur in the form of DOC, POC or gaseous
1412 C (CO₂ or CH₄). Release of old C can be measured with radiocarbon isotopes, either on bulk OC or on
1413 compound specific biomarkers. Since permafrost and Yedoma deposits contain organic C with ages
1414 of >30,000 yr BP (e.g. Zimov et al., 2006), radiocarbon could be an excellent marker to detect
1415 change in Arctic aquatic environments.

1416 Research that includes bulk radiocarbon measurements in rivers has largely focused on DOC and
1417 POC. Generally, DOC in larger river systems tends to be young. In large Arctic rivers, Amon et al.
1418 (2012) and Guo et al. (2007) measured ¹⁴C-DOC values ranging from 83 to 113% modern (1,440 to
1419 modern yr BP). Within large Arctic river basins there is significant spatial variability in riverine ¹⁴C-
1420 DOC values, reflecting dominant water and carbon source materials (Alaska, Aiken et al., 2014;
1421 O'Donnell et al., 2014). Export of contemporary DOC in rivers dominates the spring freshet, a time
1422 of year when the majority of water and DOC export occurs. In the Ob', Yenisey, Lena, Mackenzie, and
1423 Yukon rivers, Raymond et al. (2007) estimated that ~90% of DOC exported at this time was less
1424 than 20 years old. Later in the summer, DOC showed slight aging, (675 yr BP, NE Siberia, Neff et al.,
1425 2006) which is likely related to OC input from deeper active layer thaw. Winter flow is most ¹⁴C-
1426 depleted, although there can be significant variation within a large river basin even in winter.
1427 O'Donnell et al. (2014) measured winter radiocarbon ages ranging from 35 to 445 yr BP, likely
1428 related to regional groundwater travel times.

1429 Whilst major Arctic rivers mostly seem to export large amounts of young semi-labile DOC, there are
1430 also examples of mobilization of old DOC, particularly within smaller systems. In the Sagavanirktok,
1431 River, draining north Alaskan tundra, DOC age was 2,170-4,950 yr BP (Guo et al., 2007). Soil organic
1432 matter in the river basin was of similar age, and likely released old DOC from the active layer and
1433 through soil cryoturbation. The oldest DOC ever dated in surface waters is from small (first-order)

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1465 sediment-rich thaw streams draining directly into the Kolyma River, Siberia (Vonk et al., 2013;
 1466 Spencer et al., 2015). This Pleistocene-aged (>21,000 yr BP) DOC is being mobilized from old
 1467 Yedoma deposits (aged up to 45,000 yr BP) into small DOC-rich streams. This old DOC also shows
 1468 very high biodegradation potential (Vonk et al., 2013; Spencer et al., 2015; section 3.1), indicating
 1469 that it may be degraded to CO₂ well before reaching the mouth of larger river systems. (Kolyma
 1470 River, Mann et al., 2015). Deep groundwater can also be a source of ancient DOC in large river
 1471 systems (Yukon River, Aiken et al., 2014).

1472 There is abundant evidence of mobilization of old POC into Arctic lakes (e.g., Canada, Abbott and
 1473 Stafford, 1996), rivers, and estuarine sediments associated with permafrost thaw, bank erosion,
 1474 and transport of organic C. For example, Guo et al. (2007) reported ¹⁴C ages of sediment and
 1475 suspended POC in large North-American Arctic rivers of 4430-7970 yr BP and concluded that POC
 1476 release and age would increase in Arctic river systems subject to global warming. Additionally,
 1477 there has been some work in smaller coastal watersheds in the Canadian High Arctic by Lamoureux
 1478 and Lafrenière (2014) who concluded that recent permafrost disturbance delivered old (up to 6740
 1479 yr BP) POC to the aquatic system. Yedoma-derived Pleistocene aged POC has also been identified in
 1480 sediments from the Colville River Delta, which drains into the west Beaufort Sea (Schreiner et al.,
 1481 2014; 10,000-16,000 yr BP), and in thaw streams draining Yedoma deposits, Siberia (Vonk et al.,
 1482 2013; 19,000-38,000 yr BP). One caveat to studies in coastal or estuarine settings is that marine
 1483 sediments and microfossils could potentially influence the ¹⁴C age of particulate material.

1484 Contrary to bulk measurements, compound-specific biomarkers can provide more source-specific
 1485 information, and avoid many of the issues related to ¹⁴C dating of bulk OC. Spencer et al. (2008)
 1486 found elevated lignin C-normalized yields during the spring freshet across the Yukon River basin,
 1487 identifying surface vegetation as strong DOC sources. Amon et al. (2012) similarly found that
 1488 biomarker abundance changed in six of the largest Arctic rivers according to season, with high
 1489 concentrations of lignin phenols in the spring freshet (indicative of fresh vegetation) and elevated
 1490 levels of p-hydroxybenzenes during the low flow season (indicative of moss and peat-derived OM).
 1491 Concentration differences in source-tracing organic molecules, namely ¹⁴C-young, vascular plant-
 1492 derived lignin phenols and ¹⁴C-old permafrost-derived waxy lipids, were found to show a
 1493 relationship between ¹⁴C age and permafrost coverage (Feng et al., 2013). Drainage basins
 1494 associated with increasing amounts of discontinuous permafrost were characterized by older OC,
 1495 released from deeper conduits in the watershed. Likewise Gustafsson et al. (2011) found that the
 1496 average age of n-alkanes in estuarine sediments increased (1140 to 6400 yr BP) from east to west
 1497 across the Siberian Arctic, consistent with warmer climatic conditions and more discontinuous
 1498 permafrost towards the west. Additional biomarkers such as membrane lipids (ex. glycerol dialkyl
 1499 glycerol tetraethers, GDGTs; bacteriohopanepolyols, BHPs; and intact polar membrane lipids, IPLs)
 1500 may have the potential to trace terrigenous OC stored in permafrost and remobilized along Arctic
 1501 land-river-ocean transects (Rethemeyer et al., 2010; Doğrul Selver et al., 2012, 2015).

1502 4.1.2 Release of old permafrost OC as greenhouse gases

1503 The consequence of permafrost thaw beneath and adjacent to thermokarst lakes, wetlands, and
 1504 streams is the potential mobilization and return of old carbon to the atmosphere. Schaefer et al.
 1505 (2014) defined the permafrost carbon feedback as amplification of anthropogenic warming due to

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Deleted: include deep groundwater and, in some basins, glacial meltwater (Aiken et al., 2014). The DOC concentrations in these source waters, however, are typically low

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1530 carbon emissions from thawing permafrost. Direct evidence for a positive permafrost carbon
 1531 feedback to climate in thermokarst lakes is found in the radiocarbon ages and deuterium values of
 1532 CH₄ in bubbles and in spatial patterns of CH₄ emissions in thermokarst lakes. Zimov et al. (1997)
 1533 first revealed that methanogenesis in deep, cold thaw bulbs where Pleistocene-aged yedoma is
 1534 thawing beneath lakes in Siberia leads to the release of Pleistocene-aged CH₄. The release of
 1535 permafrost-derived carbon to the atmosphere in ¹⁴C-depleted CH₄-rich bubbles contributes to
 1536 climate warming, which in turn causes permafrost to thaw and more CH₄ to be produced in a
 1537 positive feedback cycle (Walter et al., 2006). Field observations and modelling showed that
 1538 permafrost-derived CH₄ emissions were highest along thermokarst margins in Siberian and Alaskan
 1539 lakes, in younger stages of lake development where permafrost thaw is most active, and in small
 1540 early-stage permafrost thaw depressions (Walter et al., 2006, 2007; Desyatkin et al., 2009; Kessler
 1541 et al., 2012; Shirokova et al., 2013). This permafrost carbon feedback was affirmed in Alaskan
 1542 thermokarst lakes by independent evidence from deuterium. Walter et al. (2008) and Brosius et al.
 1543 (2012) found that δD values of ebullition CH₄ in yedoma-type lakes in Alaska and Siberia reflected
 1544 CH₄ formation from Pleistocene-origin melt water, which has a highly negative isotopic signature.
 1545 In contrast, bubbles emitted from the centres of older yedoma lakes where permafrost is no longer
 1546 thawing (Alaska, Kessler et al., 2012), and from non-yedoma lakes, contained higher δD-CH₄ values
 1547 and younger ¹⁴C-CH₄ ages, pointing to Holocene-aged meteoric water and carbon as the substrates
 1548 for methanogenesis (Alaska, Brosius et al., 2012). On the other hand, recent work on an eastern
 1549 Canadian thermokarst lake (Holocene deposits) shows a different trend, where ebullition CH₄
 1550 emitted from the lakeshore was younger (~1550 yr BP) and had a more negative δD-CH₄ than from
 1551 the lake centre (~3250 yr BP; Bouchard et al., 2015). Most interestingly, smaller thermokarst ponds
 1552 at the same site emitted modern CH₄ even though they are exposed to peat slumping and erosion
 1553 down at least to the active layer (base of active layer ~2,200 to 2,500 yr BP) with δD-CH₄ reaching
 1554 down to -448 ‰ (Bouchard et al., 2015).

1555 It is important to note that the present-day permafrost carbon feedback from thermokarst lakes to
 1556 climate warming is likely smaller than it was in the early Holocene when thermokarst lakes first
 1557 formed on the permafrost landscape (Walter et al., 2007; Brosius et al., 2012). Walter Anthony et al.
 1558 (2014) estimated rates of carbon loss from yedoma-type lakes (in North Siberia, Alaska and
 1559 northwest Canada) to the atmosphere from 20 ky ago to the present. Their results indicate
 1560 widespread lake formation between 14-9 ky ago, generating a major northern source of ¹⁴C-
 1561 depleted atmospheric CH₄ during deglaciation. The subsequent slow-down of first-generation
 1562 thermokarst-lake formation throughout the Holocene combined with the acceleration of other
 1563 negative feedback processes (e.g. carbon sequestration by lakes, see next section) results in lower
 1564 present-day CH₄ emissions, a smaller permafrost carbon feedback, and a net negative radiative
 1565 forcing of carbon exchange between lakes and the atmosphere on climate.

1566 4.2 Carbon burial

1567 4.2.1 Carbon burial in Arctic aquatic ecosystems

1568 Inland waters receive large quantities of organic matter from their watersheds, but, globally, less
 1569 than half of this carbon reaches the ocean (Battin et al., 2009). The loss en route is attributed to (i)

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1578 mineralization to CO₂ and CH₄ (see section 3), and (ii) and sequestration into sediments of lakes
 1579 and reservoirs (Cole et al., 2007). Sediment sequestration of carbon can be substantial in relatively
 1580 lake-rich boreal and [Arctic](#) landscapes (Lehner and Döll, 2004; Fig. 1), but still receives little
 1581 attention.

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1582 Generally, total carbon mineralization rates exceed carbon burial (Battin et al., 2009; Tranvik et al.,
 1583 2009), but there are some exceptions, for example in the case of deep thermokarst lakes (see sec.
 1584 4.2.2). Lake shape is a key regulator [of](#) carbon burial; small and deep lakes ([boreal Finland](#),
 1585 [Kortelainen et al., 2004](#); [northern Québec](#), Ferland et al., 2012) bury carbon more efficiently than
 1586 large and shallow lakes. This is explained by a higher benthic metabolic capacity to process
 1587 incoming carbon and greater particle resuspension in large, shallow, and thus well-mixed lakes.
 1588 Prior to burial, degradation occurs in the water column and the uppermost sediment layers. This
 1589 can be substantial with, for example, averages up to 75% of the OC mineralized over the first few
 1590 decades following sediment deposition in boreal lakes in Québec (Ferland et al., 2014). Long-term
 1591 [\(century-scale to full Holocene\)](#) accumulation rates in sediments of [Arctic](#) and boreal non-
 1592 thermokarst lakes ranged between 0.2 and 13 g C/m²/yr across sites in Greenland, boreal Québec,
 1593 and boreal Finland (Anderson et al., 2009b; Ferland et al., 2014; Kortelainen et al., 2004; Sobek et
 1594 al., 2014). Thermokarst lakes in yedoma regions, however, show much larger long-term sediment
 1595 accumulation rates (47±10 g C/m²/yr; Walter Anthony et al., 2014; see section 4.2.2).

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1596 Similarly, coastal shelf regions bordering yedoma-rich Eastern Siberia receive rather large amounts
 1597 of carbon with accumulation rates of 36±17 g C/m² annually (Vonk et al., 2012b). Long-term
 1598 (Holocene) carbon accumulation rates in this region, however, vary between 0.1 and 2.7 g C/m²/yr
 1599 [\(Stein and Fahl, 2000; Bauch et al., 2001\)](#) suggesting significant decomposition in the sediments
 1600 after deposition and/or increases in recent accumulation rates. Furthermore, recent studies in this
 1601 region suggest that permafrost-derived carbon is preferentially buried, when compared with
 1602 marine or modern terrestrial carbon ([Siberian shelf](#), Vonk et al., 2014). This appears to [contrast](#)
 1603 with high initial biodegradability of (yedoma) permafrost DOC upon aquatic release (see section
 1604 3.1). We hypothesize that this apparent contrast can be explained by the parallel thaw-release of
 1605 different pools of organic matter in permafrost (Vonk et al., 2010; Karlsson et al., 2011). On the one
 1606 hand, DOC and buoyant, non-mineral bound POC are released that are highly sensitive to
 1607 biodegradation (e.g. [NE Siberia](#), [Vonk et al., 2013](#); [Alaska](#), Abbott et al., 2014) leading to rapid
 1608 removal in aquatic systems (Spencer et al., 2015), whereas mineral-bound, ballasted POC is
 1609 resistant to degradation and preferentially transported to (and buried in) coastal shelf sediments
 1610 ([Karlsson et al., 2011](#); [Vonk et al., 2011](#)).

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1611 4.2.2 Carbon burial in yedoma thermokarst lakes

1612 Since the last deglaciation (the past 14.7 ky), about 70% of all yedoma deposits has thawed through
 1613 the formation of thermokarst lakes and streams (Strauss et al., 2013). This has released GHG to the
 1614 atmosphere and OC to lake basin sediments and downstream export. Formation of thermokarst
 1615 systems, however, [has](#) also caused atmospheric CO₂ to be absorbed through contemporary plant
 1616 photosynthesis, senescence, and burial. While initial thermokarst basin formation caused
 1617 significant efflux of CO₂ and CH₄, as these basins evolved, nutrient-rich sediments facilitated
 1618 terrestrial and aquatic plant proliferation, leading to sequestration of OC in sediments of drained

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lake basins during the Holocene. The long-term organic carbon accumulation rate in deep, yedoma thermokarst lakes was found to be on average five times higher than in other northern lakes throughout the Holocene (Walter Anthony et al., 2014). The anomalously high carbon sequestration in yedoma thermokarst lakes was attributed to (a) thermokarst-related shore erosion and deposition of terrestrial organic matter in lake bottoms; (b) high aquatic productivity enhanced by nutrient supply from thawing yedoma; and (c) unique preservation conditions in deep thermokarst lakes.

Since GHG emissions and carbon sequestration have counteractive effects on climate (warming vs. cooling, respectively), the radiative impacts of both processes must be upscaled to understand their overall impact over long time scales. Walter Anthony et al. (2014) developed trajectories of thermokarst-basin carbon flux (for yedoma landscapes) from the last major glaciation to present, based on estimates of contemporary CH₄ flux, total yedoma carbon lost as CO₂ and CH₄, total accumulated carbon, and thermokarst-lake initiation dates. Model results indicated that yedoma thermokarst lakes caused a net climate warming at the peak of their formation during deglaciation, driven primarily by CH₄ release from thawed, decaying yedoma. However, high carbon accumulation in existing basins and a slowdown of lake formation caused thermokarst lake impact on climate to switch from net warming to net cooling around 5 ky ago, such that these basins are now net GHG sinks. Notably, long-term trends in the climate feedback potential of non-yedoma thermokarst lakes **and ponds** have not yet been extensively investigated, despite the fact that non-yedoma permafrost stores 75% of the global carbon permafrost pool (Zimov et al., 2006; Schuur et al., 2015).

4.3 Ecosystem structure and function

While there is a growing body of literature quantifying the nature, timing, and extent of permafrost thaw in the Arctic, there is considerably less literature on the direct effects of permafrost thaw on the structure and function of aquatic ecosystems, especially streams. The likelihood that permafrost thaw will substantially affect major ecological functions (e.g., photosynthesis, respiration, nutrient uptake) or food web characteristics (e.g., benthic algal biomass, macroinvertebrate community structure) is dependent on several factors, most notably the intensity, spatial extent, temporal duration, and hydrological connectivity of the disturbance associated with permafrost thaw. Here, we provide an overarching review of the potential effects of permafrost thaw on aquatic ecosystem structure and function.

4.3.1 Lakes

Arctic regions contain numerous lakes with large differences in abiotic and biotic conditions (Hamilton et al., 2001; Rautio et al., 2011), suggesting that the consequences of permafrost thaw on ecosystem function are likely to vary across **systems**. Thawing permafrost and associated changes in export of nutrients and DOM is expected to have pronounced effects on **the productivity and food web dynamics of** recipient lake ecosystems. Input of nutrients per se will **increase** primary production (mainly via effects on pelagic algae) (e.g., Alaska, Levine and Whalen, 2001 **and** O'Brien et al., 2005), **yet** input of sediments and DOM **will decrease** primary production if it leads to suboptimal conditions for photosynthesis, mainly affecting benthic algae but also planktonic algae

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when lakes are very turbid ([northern Sweden](#), Ask et al., 2009; [northern Québec](#), Roiha et al., 2015). In regions where thaw increases *both* nutrients and DOM we may expect stimulation of total primary production in clear and shallow lakes but suppression of primary production in more coloured or deeper lakes ([Sweden and Alaska](#), Seekell et al., 2015). In regions where retrogressive thaw slumping delivers mineral-rich sediments to lakes ([Mackenzie delta region](#), Thompson et al., 2008 [and Mesquita et al., 2010](#)), permafrost degradation has led to significantly greater dissolved ion content, lower DOC concentrations [following mineral adsorption](#), and increased water transparency. This has led to enhanced macrophyte development and higher abundance of benthic macroinvertebrates ([Mackenzie delta region](#), Mesquita et al., 2010 [and](#) Moquin et al., 2014) and higher abundance and diversity of periphytic diatoms ([Canadian Arctic](#), Thienpont et al., 2013). Further, DOM released following thaw is relatively labile and could support bacterial metabolism ([northern Sweden](#), Roehm et al., 2009; [NE Siberia](#), Vonk et al., 2013), resulting in increasing rates of bacterial respiration and production relative to primary production ([northern Québec](#) Breton et al., 2009 [and](#) [Roiha et al., 2015](#); [sub-Arctic Sweden](#), Karlsson et al., 2010).

These changes at the base of the food web are expected to result in a shift in the relative importance of different OC resources supporting higher consumers, by decreasing the importance of benthic algae and increasing the reliance on pelagic and terrestrial resources with increasing DOM. Heterotrophic bacteria transfer DOM to mixotrophic algae and heterotrophic protozoans, to zooplankton and zoobenthos feeding on bacteria, and via predation to higher trophic levels (Jansson et al., 2007). Another consequence of thaw and increased DOM export is an increasing degree of net heterotrophy, i.e., a decrease in the ratio between gross primary production and community respiration ([northern Sweden](#), Ask et al., 2012). Heterotrophic bacteria benefit from fresh and high carbon inputs from the catchment and the high nutrient concentrations below the thermocline ([northern Québec](#), Breton et al., 2009, Roiha et al., 2015). Respiration rates can be very high in thermokarst lakes, [favouring](#) rapid oxygen depletion and prolonged anoxia ([also northern Québec](#), Deshpande et al., 2015). This has implications for GHG production and exchange with the atmosphere (see section 3.5). Oxygen depletion following permafrost thaw may also affect the resource supply and abundance of higher consumer populations. CH₄-oxidizing bacteria, relatively abundant in many stratified thermokarst lakes (northern Québec, Crevecoeur et al., 2015), may play an important role in the carbon transfer through the food web. These bacteria are known to occur in environments where both oxygen and CH₄ are available, and they have been suggested to contribute to the zooplankton diet (Jones, 2000; northern Finland, Kankaala et al., 2006). Permafrost thaw may stimulate this C pathway but it is not clear if this could override the likely negative effect on higher consumers by oxygen depletion (Craig et al., 2015; Karlsson et al., 2015)

4.3.2 Streams

In section 2.2 we describe how permafrost thaw is likely to affect the delivery of sediment and nutrients to [aquatic](#) ecosystems. Sediment and nutrient concentrations are two of the fundamental factors that influence the structure and function of streams. However, as in lakes, there is a trade-off between the negative effects (smothering, shading, and scouring) caused by elevated levels of sediment loading versus the positive effects (fertilization) caused by elevated concentrations of soluble nutrients. Locally, and over the years-to-decades timescale of TEF disturbance, if these

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1752 | [features](#) intersect a stream they are likely to have significant impacts caused by massive inputs of
 1753 | sediment. While in some systems, TEFs have impacts that can be seen over broad catchment scales
 1754 | (e.g., [NW Canada](#), Kokelj et al., 2013), in others, the long-term and regionally-averaged effect of
 1755 | TEFs on suspended sediments may be relatively small (see section 2.2). At the same time, ongoing
 1756 | active layer deepening (and long-term, deep impacts from older TEFs) may add low levels of
 1757 | nutrients to [Arctic](#) streams over longer time scales. Currently there is very little literature to
 1758 | support these potential impacts.

1759 | Even subtle increases in the loading of limiting nutrients can have profound impacts on highly-
 1760 | oligotrophic, [Arctic](#) stream ecosystems. A whole-ecosystem nutrient fertilization experiment on the
 1761 | Kuparuk River has shown that long-term (30 year), low-level increases in soluble reactive
 1762 | phosphorus alone can have important influences on benthic autotrophic and macroinvertebrate
 1763 | community structure and can significantly increase primary and secondary production ([Peterson et](#)
 1764 | [al., 1985](#); Bowden et al., 1994; Cappelletti, 2006). On the other hand, sediment loading may offset
 1765 | the stimulatory effects of introduced nutrients and interfere with benthic stream structure and
 1766 | function. Adverse effects of sediment influx to streams include damage to primary producers,
 1767 | especially from scour during storms, which can reduce primary production and ecosystem
 1768 | respiration. Increased sediment loading may clog the interstices among streambed particles, which
 1769 | could reduce the connectivity between the hyporheic zone and surface waters, interfering with
 1770 | exchange of nutrients and dissolved oxygen (Kasahara and Hill, 2006). Sediment loading may also
 1771 | lead to instability on the stream bottom, affecting the ability of benthic macroinvertebrates to
 1772 | establish and feed (Uehlinger and Naegeli, 1998).

1773 | Recent studies have evaluated the higher order effects of sediment and nutrient loading from
 1774 | thermokarst and detected significant impacts on some aspects of the biological function of receiving
 1775 | waters. Daily rates of riverine production and respiration decreased by 63% and 68%, respectively,
 1776 | in the Selawik River in northwest Alaska in response to elevated turbidity levels that increased by
 1777 | several orders of magnitude below a massive thaw slump (Calhoun, 2012). Larouche et al.
 1778 | (submitted) studied biogeochemical characteristics of a tundra stream on the North Slope, Alaska
 1779 | over a period of three summer seasons (2009-2011) after a gully thermokarst feature impacted this
 1780 | stream several years earlier (2005). They found that 4-6 years after the initial disturbance the TEF
 1781 | still caused modest increases in the loading of sediment and dissolved solutes. Furthermore, rates
 1782 | of ecosystem production and respiration and benthic chlorophyll-a in the impacted reach of this
 1783 | stream, were significantly lower during the driest of the three summers. Rates of ammonium and
 1784 | soluble reactive phosphorus uptake were consistently lower in the impacted reach.

1785 | Benthic macroinvertebrates are typically the dominant vector of energy flow in [stream ecosystems](#),
 1786 | connecting primary production to higher trophic levels (e.g., Hynes, 1970; Merritt et al., 2008).
 1787 | [Species diversity of macroinvertebrate communities has been shown to be](#) sensitive to minor
 1788 | disturbances [in running waters](#) (Lake, 2000) [while](#) allochthonous sediment has been shown to
 1789 | significantly impact habitat composition, leading to profound effects on the distribution of
 1790 | individual organisms ([Alaska](#), Parker and Huryn, 2006). [As discussed above, lakes affected by thaw](#)
 1791 | [slumps in the Canadian Arctic \(Mackenzie delta region\) have been shown to have significantly](#)
 1792 | [greater dissolved ion content, lower DOC concentrations and increased water transparency](#)

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(Thompson et al., 2008), that have in-turn led to enhanced macrophyte development and higher abundance of benthic macroinvertebrates (Mesquita et al., 2010; Moquin et al., 2014) and higher abundance and diversity of periphytic diatoms (Thienpont et al., 2013). We are not aware of any published studies that document impacts of thermo-erosional events on the benthic community structure of Arctic stream ecosystems. In the study conducted by Larouche et al. (submitted) initial macroinvertebrate richness and diversity in Alaska was low but increased late in the season (August). Overall, the shifts in stream community structure were subtle.

It appears that Arctic headwater streams may be resilient and regain considerable functionality as local disturbance features begin to repair, particularly in the case of smaller features such as gully thermokarst disturbance that experience stabilization by re-vegetation (Jorgenson et al., 2006). While the acute impacts of slumping are obvious and notable, it is the chronic impacts of long-term, elevated nutrient and sediment loading that are of greater interest. There is growing evidence that subtle differences in sediment and nutrient delivery to Arctic headwater streams can still be apparent many years after disturbance, and that thermokarst slumping may significantly affect primary producer biomass, benthic organic nutrients, benthic invertebrate community structure and key ecosystem functions such as whole-stream metabolism and nutrient update. Averaged over thaw-impacted landscapes as a whole, these effects may often be subtle (e.g., Alaska, Larouche et al., submitted). However, it is less clear how long-term nutrient and sediment loading will change in thaw-impacted stream systems (Frey and McClelland, 2009; Lewis et al., 2012; Lafrenière and Lamoureux, 2013; Malone et al., 2013).

4.4. Export to ocean

Permafrost degradation can lead to clear changes in the biogeochemical flux of constituents from land to water (see section 2.2). Beyond the immediate site of impact, however, this constituent flux can be expected to have effects that range well into downstream environments. In many cases, changes on land can have clear impacts on coastal ocean processes, as has been shown within the estuarine zones of many large, southern rivers (Bianchi and Allison, 2009). In permafrost-impacted systems, understanding how changes in constituent flux at the terrestrial-aquatic interface will translate to changing export to the Arctic Ocean is still a challenging task. In this section, we describe the potential effects of permafrost thaw on land to ocean constituent flux, highlighting some major knowledge gaps in our current understanding of this process.

Scaling changes that are being observed at the small catchment scale to changes in ocean-bound transport requires an understanding of the rates of deposition, uptake, and decomposition of various thaw-released constituents, both in absolute terms and relative to their non-permafrost derived counterparts. These rates will vary considerably among constituent types. For example, nutrients released as a result of permafrost thaw may be taken up rapidly following release to aquatic environments (Alaska, Bowden et al., 2008), while major ions released following the exposure of mineral soils are largely conservative, and can be detected far downstream (NW Canada, Kokelj et al., 2013 and, Malone et al., 2013). Similarly, in some regions, sediment pulses associated with thermokarst disturbances decline markedly with movement downstream (Alaska, Bowden et al., 2008), while in others, thaw-associated sediment signatures are elevated across broad, catchment-wide scales (NW Canada, Kokelj et al., 2013). The signature of permafrost thaw-

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Deleted: Shifts in community structure can indicate event severity, given that benthic macroinvertebrate diversity and overall community composition are strongly related to stream ecosystem structure and function (Carter et al., 2006; Vannote et al., 1980).

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Deleted: Certainly, there is the possibility of large and important shifts in community structure and function if measured changes in sediment and nutrient flux do occur.

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1867 origin sediments and particulate organics has also been detected at the mouths of several large,
 1868 Arctic rivers (Guo et al., 2007; Gustafsson et al., 2011), and in increasing concentrations in some
 1869 Arctic coastal sediments (Feng et al., 2013).

1870 Where direct observational evidence does exist to indicate changing biogeochemical flux at the
 1871 mouths of Arctic rivers, it appears that some constituents are changing relatively synchronously
 1872 across Arctic regions, while others may show significant regional differences in their trends. For
 1873 example, flow-weighted bicarbonate fluxes appear to be increasing modestly in some Arctic
 1874 regions, such as at the mouth of the Yukon River (Striegl et al., 2005). This trend is further
 1875 corroborated by studies examining variation in riverine bicarbonate flux across Arctic watersheds
 1876 with differing permafrost coverage (Tank et al., 2012a; Tank et al., 2012c; Pokrovsky et al., 2015).
 1877 In contrast, the downstream transport of DOC to coastal areas appears to be increasing in some, but
 1878 decreasing in other, regions, based both on direct river-mouth measurements over time and sub-
 1879 watershed comparisons across permafrost gradients (Kawahigashi et al., 2004; Striegl et al., 2005;
 1880 Tank et al., 2012a). Furthermore, DOC originating in old permafrost may be preferentially degraded
 1881 within stream networks, and thus may not be detectable at the river mouth (Yedoma-rich eastern
 1882 Siberia; Spencer et al., 2015). It therefore remains difficult to attribute the few documented changes
 1883 at the mouths of Arctic rivers to either up-catchment permafrost degradation or, for example, to the
 1884 more widespread effects of changing temperature and precipitation patterns. For example, fluxes of
 1885 DOC will be affected by changes in the composition and overall production and decomposition of
 1886 vegetation (e.g., across northern regions, Laudon et al., 2012), in addition to the exposure of organic
 1887 soils via permafrost thaw. Similarly, fluxes of bicarbonate will be affected not only by the thaw-
 1888 mediated exposure of deeper mineral soils, but also by increases in root respiration that affect
 1889 weathering rates (Beaulieu et al., 2012). For many constituents, changes in the seasonality of
 1890 precipitation may also affect constituent flux and concentration (northern Canada, Spence et al.,
 1891 2011). Research to explore how permafrost thaw affects aquatic biogeochemistry across nested
 1892 spatial scales, and to further elucidate the mechanisms of changing chemistry at scales where
 1893 direct, mechanistic, observations are possible, will greatly aid our ability to understand how up-
 1894 catchment permafrost degradation affects biogeochemical flux to the coastal ocean.

1895 The effect of changing riverine fluxes on coastal ocean processes cannot be considered without also
 1896 considering thaw-induced changes in coastal erosion. Rates of coastal erosion vary by region, in
 1897 part because of regional differences in ground ice content and bluff height (Lantuit et al., 2013).
 1898 Overall, however, the impact of coastal erosion on biogeochemical flux to the ocean appears to be
 1899 significant. Estimates of carbon release by coastal erosion vary significantly, and range between 5-
 1900 14 Tg OC/year for the entire Arctic combined (Rachold et al., 2004; Jorgenson and Brown, 2005;
 1901 Vasiliev et al., 2005; Couture, 2010; Vonk et al., 2012b) with highest delivery rates in the Laptev and
 1902 East Siberian Seas (Vonk et al., 2012b). This value is nearly a third of the combined, circum-Arctic,
 1903 estimated delivery of DOC (34 Tg; Holmes et al., 2012) and POC (6 Tg; McGuire et al., 2009) via
 1904 rivers each year. Rates of coastal erosion, however, appear to be increasing both in the Russian and
 1905 the Alaskan Arctic (Jones et al., 2009; Günther et al., 2013) which is due to decreasing sea ice
 1906 content, allowing for higher storm intensity and wave fetch, along with increasing summertime sea
 1907 surface temperature and a rising sea (IPCC, 2013).

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Documented changes in riverine biogeochemistry and coastal erosion rates may in turn have a significant effect on carbon and nutrient cycles in the near shore ocean. Where the delivery of DOC and POC increase, the attenuation of light will reduce photosynthetic uptake of CO₂ (NW Canada, Retamal et al., 2008). Concomittantly, increases in both bacterial production, and the decomposition of this organic matter to carbon dioxide CO₂ may occur. On the East Siberian Shelf, for example, large zones of CO₂ outgassing have been shown to occur alongside plumes of DOC that have a clear terrestrial isotopic signature (Anderson et al., 2009a). In addition, increases in light caused by sea ice retreat could also increase photochemical degradation of riverine DOM to CO₂ (e.g., Tank et al., 2012b). If changing delivery of organic matter does affect coastal CO₂ saturation, this could combine with changes in bicarbonate flux to have a significant impact on nearshore aragonite saturation, compounding the effects of temperature and sea ice melt on ocean acidification in the Arctic (Steinacher et al., 2009; Yamamoto-Kawai et al., 2009). In much of the Russian Arctic, organic carbon transport from land to ocean is already high (Holmes et al., 2012; Vonk et al., 2012b) when compared to the North American Arctic, and rivers are relatively bicarbonate-poor (Tank et al., 2012c). Thus, increasing delivery of organics to coastal zones, and the resultant CO₂ production, could further reduce aragonite saturation in a region where near shore regions are already poorly buffered (Anderson et al., 2011; Tank et al., 2012c). For these, and other constituents such as nutrients, we still have much to learn about how fluxes to the coastal ocean are changing, and how this change may affect near shore biogeochemical function (Tank et al., 2012b; Le Fouest et al., 2013; Letscher et al., 2013). Understanding the effect of these changing fluxes in general, and the specific importance of permafrost thaw for changing biogeochemistry in the coastal Arctic, remains a clear priority for future research.

5. Summary, feedbacks, and future research needs

Permafrost thaw has a broad range of effects on the functioning of aquatic ecosystems. These include changes in optical and thermal properties, altered chemistry of the water column and of sediments, changes in contaminant loads, and altered potential for bio- and photo-degradation, which in turn affect gas fluxes, carbon burial and export of thaw-released constituents downstream and to the coastal ocean. In addition, these changes affect microbial communities and processes, primary production, and trophic structure in thaw-impacted systems.

In addition to these effects, thermokarst can be expected to initiate important feedbacks to climate change, which will often result from processes that range beyond those directly impacting aquatic systems. Below, we present some summary thoughts, discuss broad scale feedbacks to climate, and provide an assessment of future research needs based on the summary presented in this review.

5.1 Summary

Ground-ice content, topography, and soil type are the main drivers for both types of permafrost thaw (press vs. pulse) and associated release of constituents into aquatic systems: (i) When thaw is manifested as a pulse disturbance, this leads to thermokarst lakes (lowland terrain) or slumping (hillslope terrain). For both features, the soil type of the pulsed material generally determines the release and effect of constituents; a pulse of OC-rich soils will colour thermokarst lakes (affecting

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1987 stratification etc.), will cause low transparency (high turbidity) in aquatic ecosystems after thaw,
1988 and will lead to increasing OC ~~export~~. A pulse of mineral-rich soils, however, might lead to clearer
1989 thermokarst lakes and decreasing OC in the water column, due to sorption of matter to mineral
1990 surfaces. (ii) When thaw is manifested as a press disturbance, this generally leads to longer flow
1991 paths and increasing residence time in soils. Here, *the soil type of the thawed material* generally
1992 determines the release and effect of constituents; thaw of OC-rich soils will lead to higher OC ~~export~~
1993 whereas thaw of mineral-rich soils will lead to lower OC ~~export~~.

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1994 The fate of released constituents and their feedbacks to climate depends on the propensity of these
1995 constituents for degradation versus burial, which is determined by environmental parameters as
1996 well as intrinsic properties. Also, it is important to distinguish between thaw-mobilization of old
1997 permafrost OC versus contemporary OC, when considering the feedback potential. The
1998 degradability of released OC, representing the most direct carbon-climate "linkage", can be divided
1999 into a dissolved and a particulate fraction; the biodegradability of DOC is determined by source,
2000 chemical character, nutrient availability, temperature, and prior microbial and photochemical
2001 processing. Furthermore, DOC is generally more degradable when flushed from continuous
2002 permafrost regions, surface litter, active layer soils, and yedoma, but less degradable when flushed
2003 from deeper mineral soil layers. Photodegradation of DOC is relatively high in thaw-impacted
2004 systems that are shallow, rich in light-absorbing DOM, or that undergo short-term (days to weeks)
2005 stratification events. Photodegradation can however be hampered by slumping of OC-rich soils
2006 (decreasing transparency) or slumping of mineral-rich soils (adsorbing CDOM). Our understanding
2007 of degradation of POC and the factors influencing it is still remarkably poor. Burial of OC is
2008 generally lower than total OC ~~remineralisation~~ to CO₂ and CH₄, particularly in large and shallow
2009 lakes, but can be higher in small and deep lakes, thermokarst-yedoma lakes, and on the coastal
2010 shelf.

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2011 There are few studies on the effects of permafrost degradation on the ecology and food web
2012 structure of aquatic ecosystems. Thus, we can only speculate about future ecological conditions
2013 based on our current understanding of the function of high latitude systems and the effects of
2014 permafrost degradation on the physics and chemistry of impacted systems. The impact of
2015 permafrost thaw on foodwebs and ecosystem functioning appear likely to be driven by changes in
2016 the inputs of nutrients, DOM, and sediments. Primary production is stimulated by nutrient input but
2017 can be hindered by light suppression following increasing input of CDOM or OC-rich sediments,
2018 particularly when DOM concentrations are great enough that the positive effect of increasing
2019 organic nutrients is overwhelmed by the negative effect of decreasing light penetration (Seekell et
2020 al., 2015). Benthic communities can be destabilized by high sediment loading but may thrive when
2021 slumping of mineral-rich sediment leads to increasing water clarity. Overall, food web changes may
2022 lead to shifts in (i) OC resources supporting higher consumers, and (ii) the net heterotrophy of
2023 systems. Increasing terrestrial DOM input may be transferred to zooplankton and zoobenthos via
2024 increased heterotrophic bacterial production. On the other hand, in response to elevated turbidity
2025 levels, production, and respiration may decrease.

2030 5.2 Climate feedbacks

2031 Permafrost thaw enables important feedbacks to climate through activating and remobilizing
2032 previously frozen carbon pools (Schuur et al., 2015) that generate fluxes of CO₂ and CH₄ (a net
2033 climate warming) or generate increasing accumulation in sedimentary basins (a net climate
2034 cooling). There are, however, more climate feedbacks that, while not directly related to aquatic
2035 processes, do have an important effect on the functioning of these systems. For example,
2036 thermokarst-enabled increases in inundated landscape area can be followed by loss of tree cover in
2037 the area surrounding thaw depressions, and a shift towards sedge-dominated fen vegetation (e.g.
2038 Jorgenson et al., 2001). The resulting change in albedo could subsequently affect regional radiative
2039 forcing, in a direction which will depend upon the manner in which vegetation changes affect snow
2040 cover (Notaro and Liu, 2007), and the relative change in cover of peat, forest, and water, because of
2041 the differences in albedo between these land cover types (e.g. Lohila et al., 2010). Additionally,
2042 energy partitioning into latent and sensible heat fluxes may be altered significantly upon thaw, as
2043 lake surfaces would be increasingly more important in certain regions, creating a distinct
2044 microclimate with high evapotranspiration and low sensible heat flux (e.g. Rouse et al., 2005). Lake
2045 energy balance varies widely with depth (e.g. Eaton et al., 2001), adding importance to the temporal
2046 changes in thermokarst lake sizes. Finally, thermokarst-enabled changes in the emission of biogenic
2047 volatile organic compounds (BVOCs) through landscape shifts could also affect regional climate,
2048 because secondary aerosols originating from BVOCs facilitate cloud formation (e.g. Ehn et al., 2014).
2049 Taken as a whole, permafrost thaw and the occurrence of thermokarst will produce diverse and
2050 contra-directional climatic effects at different landscape, regional, and global scales. An approach
2051 that considers these multiple effects is therefore needed to understand how thermokarst feeds back
2052 to regional and global climates.

2053 5.3. Future needs for research

2054 Aquatic ecosystems are widely recognized as locations of active processing and burial of the
2055 organic matter they receive (Cole et al., 2007; Battin et al., 2009), and, lately, also receive more
2056 attention in climate-carbon interactions in the Arctic (e.g. Sobek et al., 2003; Feng et al., 2013; Vonk
2057 and Gustafsson, 2013; Olefeldt and Roulet, 2014). However, in this review we have also identified
2058 numerous gaps in our knowledge of the diverse effects of permafrost thaw on aquatic ecosystems
2059 and the consequential feedbacks on climate. We therefore make the following recommendations for
2060 future research directions, where we make the division into general directions, directions specific
2061 to streams and rivers, directions specific to thermokarst lakes, and the use of specific techniques:

2062 5.3.1 General future research directions

2063 • Fluxes and degradation of particulate OC from thawing permafrost

2064 Permafrost thaw, particularly when manifested as a pulse disturbance, can deliver substantial POC
2065 inputs to aquatic systems by exposing and rapidly thawing deep permafrost deposits. However,
2066 studies focusing on OC fluxes from thawing permafrost have to date mostly focused on DOC.
2067 Whereas DOC from collapsing permafrost is among the most biodegradable reported in natural
2068 systems (Vonk et al., 2013a; Abbott et al., 2014; Spencer et al., 2015), the biodegradability of POC,
2069 released in far larger quantities, has never been properly assessed.

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2074 • The relative mobilization and degradability of old versus contemporary carbon
 2075 As thawing permafrost increases shoreline contact between lakewater and soils, and increases
 2076 direct slumping into lake and stream systems, the carbon that is introduced to aquatic systems will
 2077 be from both shallow, contemporary, soil layers and from older, permafrost soils. Although
 2078 permafrost DOC derived from yedoma appears to be highly degradable (see section 3.1), there is
 2079 also evidence that some thermokarst lakes emit modern carbon (see section 4.1.2). Understanding
 2080 the relative susceptibility of OC pools with permafrost-origin versus contemporary-origin to bio-
 2081 and photodegradation, and the relative mobilization of these two pools as a result of permafrost
 2082 thaw across various regions and aquatic ecosystems, will help our ability to quantify feedbacks to
 2083 climate in thaw-impacted systems. The priming effects generated by, for example, light and
 2084 photosynthetic exudates on the consumption of old OC also needs to be further explored.

2085 • Influence of permafrost thaw on fluxes to coastal ocean
 2086 Our understanding of the effect of changing constituent fluxes following permafrost thaw on the
 2087 optical characteristics, primary production, and biogeochemistry of coastal Arctic systems is still
 2088 limited and remains a clear priority for future research.

2089 • Resiliency of stream ecosystems to direct thermokarst impacts
 2090 Very few studies have reported on the intensity or the duration of thermokarst impacts on the
 2091 structure of biological communities or the function of key ecological processes (e.g., photosynthesis,
 2092 respiration, and nutrient uptake) in Arctic streams (Calhoun, 2012; Larouche et al., submitted). It is
 2093 impossible, therefore, to do more than speculate about how the ecosystem services provided by
 2094 Arctic streams and rivers are changing in response to this regional impact of climate change.

2095 • Trophic structure and food web processes
 2096 The effect of permafrost thaw on aquatic autotrophic and heterotrophic communities, and their
 2097 interaction, remains poorly studied. For example, resource use and growth by consumers in
 2098 thermokarst lakes has not been quantified to date. Long-term effects of nutrient and sediment
 2099 loading in thaw-impacted stream systems are still understudied, but are vital for effects on and
 2100 shifts in receiving foodwebs.

2101 • Microbial diversity and processes
 2102 Microbial diversity studies have only recently begun, and there are many gaps in understanding.
 2103 For example, the diversity and roles of viruses, likely the biologically most abundant particles in
 2104 thaw waters, have not received attention to date. The composition of winter microbial communities
 2105 in ice-covered thermokarst lakes and ponds is at present unknown, and only minimally studied in
 2106 rivers (Crump et al., 2009) and the microbial processes operating under the ice have been little
 2107 explored. These deserve special attention, given the long duration of ice-cover in northern lakes,
 2108 and the evidence of prolonged anoxia in these waters that favour anaerobic processes such as
 2109 methanogenesis. The spring period of ice melt and partial mixing, and the prolonged period of
 2110 mixing in fall, may be important for gas exchange as well as key aerobic microbial processes such as
 2111 methanotrophy, and these transition periods also require closer study.

2112 • Improved assessment of underwater UV irradiance
 2113 To specifically understand photodegradation of both old and contemporary DOM in thaw waters
 2114 (see also bullet above), we further recommend work to (i) quantify UV spectral irradiance in thaw-
 2115 impacted water columns, (ii) understand the residence time of DOM in the UV-exposed portion of

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2130 the water column and its variability with changing mixing regimes, and (iii) identify the factors that
2131 control vertical losses of DOM and the lability of permafrost DOM to photo-degradation.

2132 5.3.2 Research directions specific to streams and rivers

2134 • Watersheds of small and intermediate size

2135 Research to date has been somewhat skewed towards large rivers basins and estuaries; small
2136 watersheds and headwater streams where processes and change are easier to elucidate remain
2137 under-studied, meaning that it is often difficult to link measurements to clear source areas or
2138 processes.

2139 • Sediment erosion versus delivery to streams

2140 Streambank erosion effectively delivers 100% of eroded sediments to streams. But TEFs that form
2141 at some distance from streams may deliver far less sediment mass, C, N, and P to streams. To scale
2142 the effects of hillslope thermokarst to aquatic systems at broad spatial scales, we must better
2143 quantify how the position of various TEF features in the landscape moderates their effect on aquatic
2144 ecosystems.

2145 • Influences of hyporheic processes

2146 It is well known that the hyporheic zone (region below and alongside the stream) contributes
2147 substantially to nutrient and carbon processing in temperate and tropical streams (e.g., Boulton et
2148 al., 1998). Recent research has shown that despite the presence of permafrost, the hyporheic zone
2149 is equally important to the ecological functions of Arctic streams (e.g., Zarnetske et al., 2008).
2150 However, we do not know how thermokarst impacts will affect hyporheic processes or vice versa
2151 (Edwardson et al., 2003).

2152 5.3.3 Research directions specific to thermokarst lakes

2154 • Thermokarst lake processes in non-yedoma systems

2155 The emission and burial of thaw-released OC in yedoma thermokarst lakes has been a relatively
2156 large focus of research attention (e.g. Walter Anthony et al., 2007; 2014; Sepulveda-Jáuregui et al.,
2157 2015), but the fate of thaw-released OC in thermokarst lakes in non-yedoma regions is still
2158 understudied. Considering that the yedoma region holds 210-456 Pg C and the total permafrost
2159 region holds 1330-1580 Pg C (Schuur et al., 2015), this makes non-yedoma regions holding 66-87%
2160 of the total permafrost C important to consider, particularly where non-yedoma regions are also
2161 lake-rich.

2162 • Physical and hydrological dynamics of thermokarst lakes

2163 Current gaps in our understanding of the physical and hydrological dynamics of thermokarst lakes
2164 include the quantification of sediment heat transfer, penetration of solar radiation through ice
2165 cover, modelling of diffusive GHG exchange in small lakes, wave and energy dynamics associated
2166 with floating ice, and the extent of groundwater flow (Kirillin et al., 2012). Furthermore, material
2167 transport caused by advective water transfer between shallow zones to bottom waters such as
2168 found in MacIntyre and Melack (1995) in other systems has not been addressed in thaw waters.

2169 • Hydrodynamic effects of high CH₄ and CO₂ concentrations

2170 Gradients in gas concentrations (particularly during ice-covered periods) can cause density
2171 differences in the water column that can modify stratification such as suggested by Deshpande et al.

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2176 (2015) but these effects have been little studied to date. Also, the effects on stratification by gas
2177 bubble trains associated with ebullition from sediments (Walter et al., 2006) have received little
2178 attention.

2179 • Role of CH₄ oxidation in thermokarst systems

2180 The emission of CH₄ from aquatic ecosystems is significantly offset by microbial oxidation of CH₄
2181 (Trotsenko and Murrell, 2008). For example, in northern lakes, up to 88% of CH₄ produced in
2182 sediments is oxidized by microbes (e.g., Bastviken et al., 2008) and abundant methanotrophs have
2183 been observed in thermokarst lakes. Oxidation of CH₄ has recently been detected through
2184 laboratory incubation studies of thermokarst lakes in the boreal and tundra zones of Alaska
2185 (Martinez-Cruz et al., 2015), however, numerous questions remain to be answered with respect to
2186 (i) the extent to which CH₄ oxidation offsets whole-lake emissions in thermokarst-lake systems, (ii)
2187 which CH₄-carbon pools are subject to oxidation (contemporary vs. old carbon), (iii) microbial
2188 community dynamics, and (iv) biogeochemical and ecological controls over CH₄ oxidation among
2189 different thermokarst lake types.

2190 • Lake carbon burial

2191 Our knowledge of the relative role of burial versus processing in northern lakes remains poor.
2192 Tranvik et al. (2009) project that carbon burial in polar lakes will decrease whereas carbon burial
2193 in boreal lakes will increase. This review, however, points out that other factors such as permafrost
2194 type (yedoma vs. non-yedoma; Walter Anthony et al., 2014) or lake shape (small and deep vs. large
2195 and shallow; Ferland et al., 2012) strongly affect burial efficiencies and may overrule the distinction
2196 between boreal and polar regions. More research is needed to shed light on these processes.

2198 **5.3.4 The use of specific techniques in future research**

2199 • Usage of high-resolution automated loggers in thermokarst lakes

2200 Many thermokarst lakes undergo rapidly cycling stratification events (i.e. diurnal or several day)
2201 that are hard to capture with sparse measurements. The increased use of high resolution,
2202 automated temperature and O₂ loggers is likely to yield new insights into short term (single to
2203 several day) stratification and mixing dynamics, even in those lakes currently considered to be well
2204 mixed in summer.

2205 • Remote sensing

2206 We recommend increasing usage of high-resolution satellite remote sensing to assess (i) local
2207 landscape conditions that affect thermokarst formation, (ii) the changing areal coverage of
2208 thermokarst lakes and ponds in both discontinuous and continuous permafrost regions, as well as
2209 (iii) changing DOC lake concentrations (e.g. Watanabe et al., 2011) derived from changing lake
2210 surface colour as a result of permafrost thaw.

2211 • Radiocarbon dating

2212 The Arctic aquatic system should provide an early and sensitive signal of change in the cycling of OC
2213 in the terrestrial environment. The development of new direct methods to date aquatic dissolved
2214 CO₂ (Billett et al., 2012; Garnett et al., 2012) has significantly increased our capacity to measure the
2215 source and age of CO₂ released from Arctic landscapes; these along with existing dating tools for
2216 POC, DOC, and CH₄, provide researchers with a strong methodological basis to quantify and detect
2217 the release of aged C into the aquatic environment. This will allow us to detect change or rates of
2218 change in areas of the Arctic undergoing differential rates of climate warming and address the key

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2228 issue of whether “old” carbon (fixed 100s or 1000s of year BP) is being released directly or
 2229 indirectly into the atmosphere.
 2230 • *Eddy correlation flux measurements on thermokarst lakes*
 2231 Given its general applicability for studying lake-atmosphere exchanges of carbon (e.g. Vesala et al.,
 2232 2012), we recommend increasing the application of eddy covariance on thermokarst lakes. We
 2233 suggest that particular attention be paid to: (i) eddy flux footprint analysis; because the flux
 2234 footprint often consists of a mixture of terrestrial and aquatic fluxes (Wille et al., 2008), it is
 2235 important to use an appropriate footprint model (Vesala et al., 2008) supplemented with localized
 2236 flux measurements (Sachs et al., 2010; Pelletier et al., 2014). (ii) usage of recently developed, low-
 2237 maintenance instrumentation, such as robust, low power, open-path and enclosed-path gas
 2238 analysers for CO₂ and CH₄ (Burba et al., 2012). (iii) the development of harmonized data processing
 2239 protocols, as past efforts to compile datasets from large, terrestrial eddy covariance networks, such
 2240 as FLUXNET (Baldocchi et al., 2001) have shown the importance of consistent data processing
 2241 protocols to ensure comparability between sites. Processing protocols should be revised for
 2242 application over lakes due to significant differences in surface processes of aquatic and terrestrial
 2243 ecosystems (e.g. Vesala et al., 2012). Given the wide range of thermokarst lake sizes and types, a
 2244 network of several flux towers has a great potential to better understand lake-atmosphere
 2245 interactions of these ecosystems.

2247 5.3.5 Inclusion and prioritization in models

2248 The export of OC and other constituents from Arctic aquatic systems remains poorly represented in
 2249 ecosystem, landscape, and permafrost models. Generally, linkages between permafrost thaw and
 2250 changes in surface hydrology (i.e. whether the Arctic landscape becomes wetter or drier after
 2251 permafrost thaw), are poorly understood (Schuur et al., 2015). In the terrestrial ecosystem model
 2252 (TEM) presented in McGuire et al. (2010), a riverine DOC export component is included which
 2253 stems from simulating production and export of DOC from land. However, processing within rivers
 2254 is not accounted for, which would ideally be needed to couple observed DOC data to input of DOC
 2255 from soils. In addition, there are many other components that are currently not included in global
 2256 or landscape-scale models, such as the release of DOC from pulse disturbances, the release and
 2257 transport of POC from permafrost thaw (by pulse and press disturbances), or the release and
 2258 transport of OC into thermokarst systems. Non-OC constituents (nutrients, sediment) are even
 2259 more poorly represented in these models. Improving aquatic constituent fluxes, processing and
 2260 transport should be a key prioritization in future model development.

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 2264 (www.cen.ulaval.ca/thaw2014/), held at the Centre d'études nordiques (CEN), Université Laval,
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 2268 in Transition' (ADAPT). We also acknowledge funding support from ADAPT towards the publication

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2309 of this article, and support to individual authors from ArcticNet, the Canada Research Chair
2310 | program, NSERC, FRQNT, the Campus Alberta Innovates Program, the PAGE21₁, and DEFROST
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2316 to thank Charlene Nielsen (University of Alberta) for creating Figure 1.

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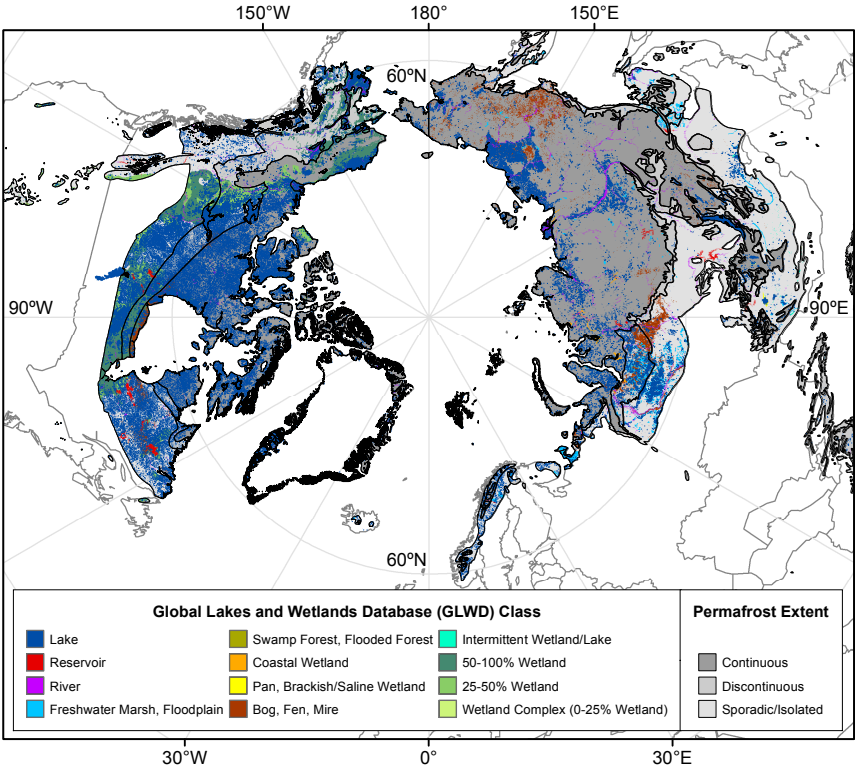
2320 **Figures**

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2322 **Figure 1:**

2323 Map of the permafrost zones in the northern hemisphere (grey scale; Brown et al., 1998)
2324 | superimposed on **water bodies** from the Global Lakes and Wetlands Database (Lehner and Döll,
2325 2004).
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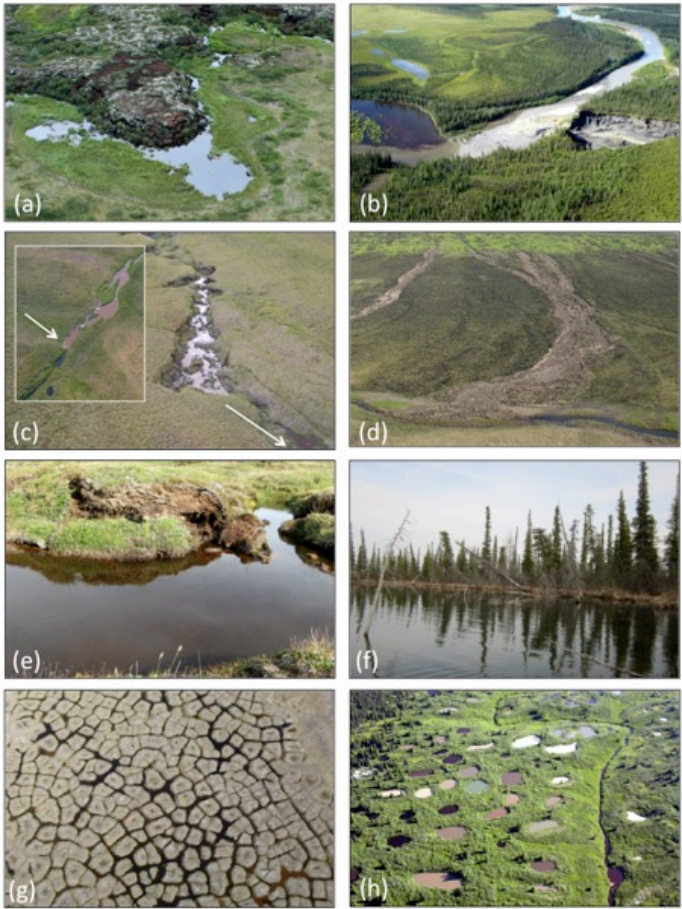
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Figure 2
Photos of typical thermokarst processes: (a) Thermokarst lake SAS1, located in a sub-Arctic peat bog near Kuujjuarapik-Whapmagoostui, Québec, Canada. The water body lies next to a thawing, organic-rich palsa (permafrost mound), and is 25 m in its maximum dimension. (b) Massive thaw slump on the Selawik River near Selawik, Alaska, US. Sediment discharge from the feature has entirely blocked the river. Note the turbidity downstream. (c) Gully thermokarst on the Toolik River, Alaska, US, and impact on receiving stream (inset), (d) active layer detachment slides near the Anaktuvuk River burn area on the North Slope of Alaska, (e) trough pond on Bylot Island, Nunavut, Canada featuring active erosion, (f) thermokarst lakes in Mackenzie Delta, Northwest Territories, Canada, showing active shoreline slumping, (g) polygonal landscape on Bylot Island, showing ice-wedge trough ponds, and (h) thermokarst lakes and ponds with a wide range in colour near Kuujjuarapik-Whapmagoostui, Québec, Canada. Photo credits: a, Bethany Deshpande; b, Ben Crosby; c, d, William Breck Bowden; e, g, and h, Isabelle Laurion; f, Suzanne Tank.



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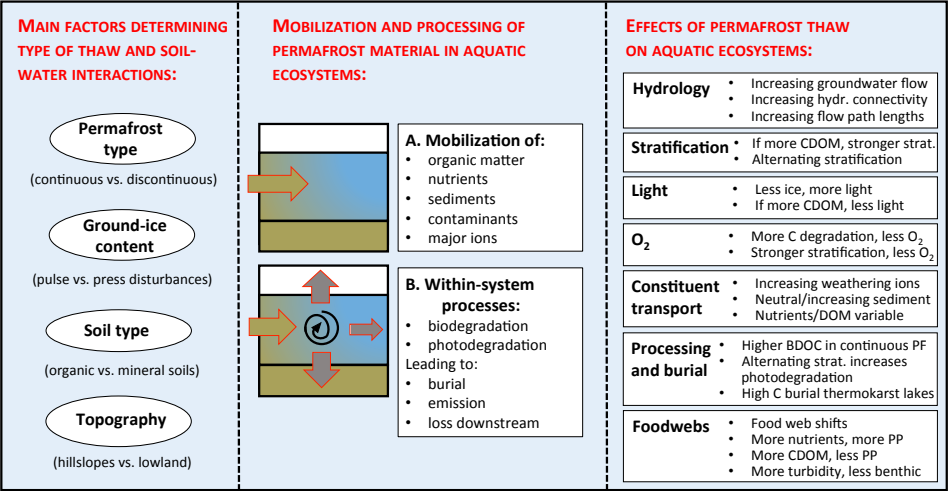
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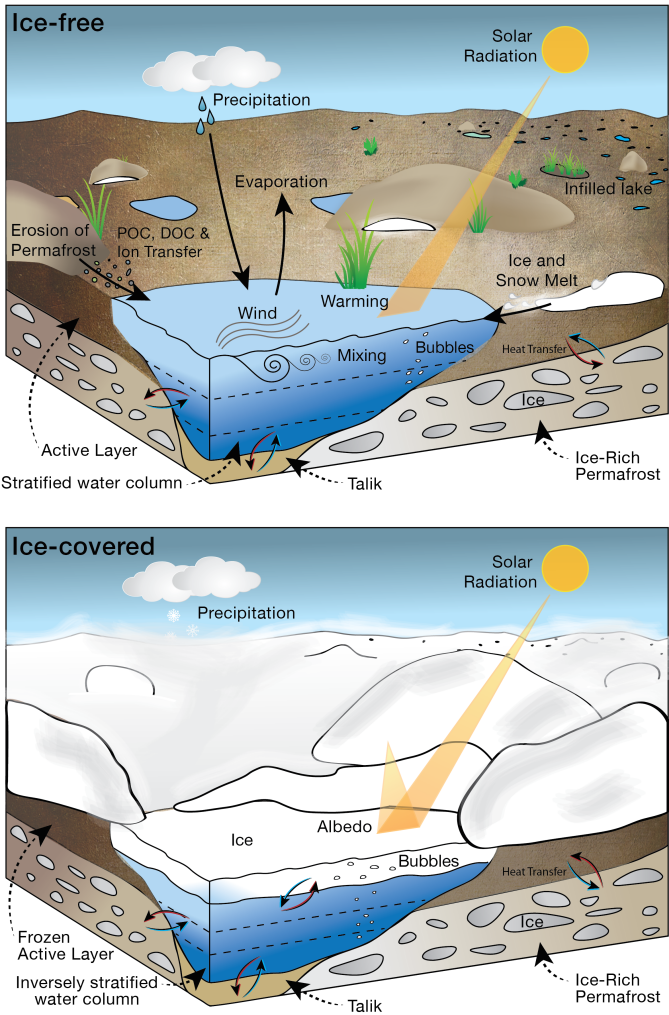
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2348 **Figure 3**
 2349 Conceptual diagram of (left) factors determining thaw type and soil-water interactions, (middle)
 2350 mobilization and processing of permafrost material into aquatic ecosystems, and (right) effects of
 2351 permafrost thaw on aquatic ecosystems.
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2354 **Figure 4**
2355 Physical limnological characteristics of permafrost thaw lakes in the ice-free and ice-covered
2356 seasons. Note that shallow ponds and lakes that freeze to the bottom in winter are not considered
2357 in this schematic.



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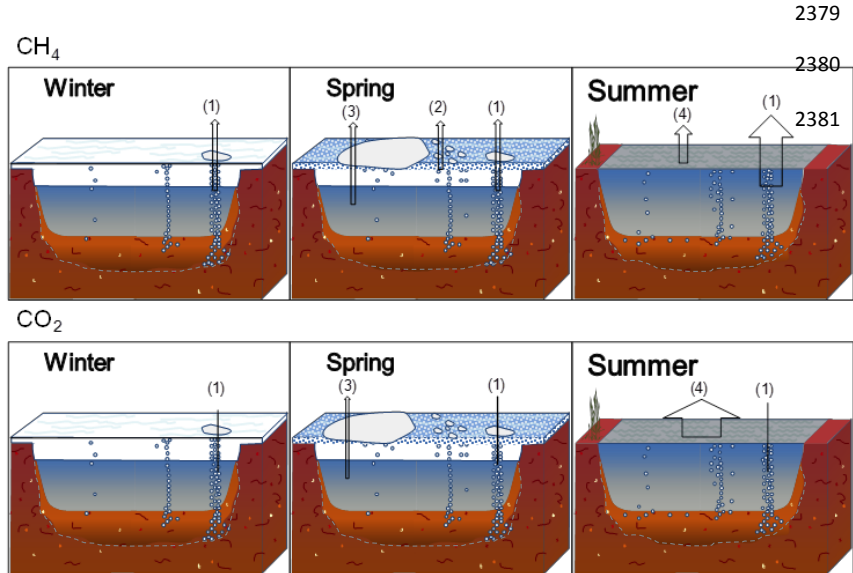
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Deleted: A talik is a region of ground that remains unfrozen throughout the year as a result of heat transfer through the water column of lakes (or other freshwater bodies). A palsa is a raised mound of permafrost found in many subarctic and arctic wetland areas.

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2369 **Figure 5**
 2370 Schematic of CH₄ and CO₂ emission pathways during different seasons in thermokarst lakes. The
 2371 thickness of arrows indicates the relative magnitude of contribution from each pathway according
 2372 to a study of 40 Alaskan lakes (Sepulveda-Jáuregui et al., 2015): (1) Direct ebullition through ice-
 2373 free hotspot seeps in winter and from all seep classes during the last month of ice cover in spring
 2374 and in summer; (2) ice-bubble storage emission during spring ice melt; (3) Storage emission of
 2375 dissolved gases accumulated under lake ice when ice melts in spring; (4) Diffusion emission from
 2376 open water in summer. The background ebullition mode, discussed in the text, is not shown. The
 2377 dashed line indicates the boundary between the thaw bulb under lakes and the surrounding
 2378 permafrost. Figure modified from Sepulveda-Jáuregui et al., 2015.



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, the lakes on yedoma-like permafrost were typically stratified in summer despite their shallow depths, with less than 0.1 mg O₂/L at the bottom of the water column.

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and lakes with a maximum depth of < 2 m tend to freeze completely

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Many thermokarst lakes are likely to be cold polymictic, undergoing stratification events that become established and then break down over diurnal, or several day, cycles. As a result, the increased use of high resolution, automated temperature loggers is likely to yield new insights into these short term stratification and mixing dynamics, even in those lakes currently considered to be well-mixed in summer. For example, in western Siberia, an air-temperature increase of 15°C from an anomalous heat wave resulted in increased surface-water temperatures of 10°C, and the formation of a strong temperature gradient in the water column (Pokrovsky et al., 2013). Additionally, there may be strong diurnal variations in stratification and mixing. On the other hand, very shallow, small thermokarst lakes (e.g., collapsed ice-wedge trough ponds; Fig. 2) can remain stratified for most of the summer, with only surface waters showing these diurnal dynamics (Negandhi et al., 2014; Bouchard et al., 2015).

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2.3 Mobilization of contaminants

Contaminants reach remote polar regions following long-range atmospheric transport and deposition, as well as through increased local waste production from marine transport and industrial or mining activities. The frozen soils of permafrost have historically been considered a barrier to the movement of contaminants and many waste and dump sites use containment strategies, which rely on the low mobility of contaminants in permafrost soils (Grannas et al., 2013). However, the warming arctic climate may lead to an increased mobility of contaminants, either stored in soils at waste sites or historically-accumulated in permafrost, into arctic surface waters (Armitage and Wania, 2013; Chételat et al., 2014).

Thawing permafrost may have major consequences for contaminant transport and transformations in the Arctic due to: a) physical changes in the hydrological cycle leading to the remobilization of contaminants from contaminated soils or sediments, b) chemical changes due to the release of nutrients and organic carbon from previously frozen soils and c) biological changes via the microbial transformation of contaminants. Most studies on the interactions between contaminants and permafrost soils have concentrated on fuel products, persistent organic pollutants (POPs) and metals such as mercury (Hg) and lead (Pb).

Permafrost thaw may cause increased mobility of contaminants from catchment soils to surface waters due to accelerated soil/peat erosion, altered hydrological flow (increasing

hydrological connectivity) and increased runoff leading to exposure of soluble contaminants (Fortier et al., 2007; Klaminder et al., 2008). Increased lateral hydraulic conductivity may accelerate the downhill movement of contaminants through the large pores, lenses and veins created in the active layer by the thawing of ice-rich permafrost (Dyke, 2001). After thaw, permafrost is no longer an impermeable barrier to contaminants, allowing for infiltration into soils and aquatic systems (Grannas et al., 2013). The reduced surface area of thermokarst lakes in some areas due to the creation of drainage channels may lead to increased contaminant concentrations in the remaining surface waters (Macdonald et al., 2005).

Traditionally, the distinction is made between (i) organic contaminants, and (ii) inorganic contaminants. Studies of *organic contaminants* (hydrocarbons or non-aqueous phase liquids) predict lateral movement in the active layer with limited vertical transport in areas of continuous permafrost (Carlson and Barnes, 2011; Dyke, 2001). Vertical migration is possible in some regions (McCarthy et al., 2004) yet permafrost acts as a low-permeability barrier in others (Curtosi et al., 2007). Organic contaminants may migrate downwards into frozen soils in areas of discontinuous permafrost due to more variable distribution (Carlson and Barnes, 2011). Few studies have yet directly focused on physical changes of permafrost on *inorganic contaminant* mobility. Studies using mass-balance calculations and paleoecological techniques have linked thermokarst erosion in peatlands and the release of Hg into lake surface waters (Klaminder et al., 2008; Rydberg et al., 2010). Stable isotope analysis also suggests that the weak recovery of Pb contamination in two sub-arctic lakes (despite dramatically reduced atmosphere inputs) may be linked to the subsidence of thawing permafrost soils (Klaminder et al., 2010). Permafrost degradation may affect the mobility of inorganic contaminants differently across different Arctic regions. One study found that lakes impacted by the development of retrogressive thaw slumps in siliciclastic soils had lower levels of Hg in surface sediments when compared to reference lakes (Deison et al., 2012). In this instance, thaw slumping may have led to a dilution of organic material and associated mercury (Hg) due to high inorganic sedimentation rates. On the other hand, a recent study on small thermokarst lakes located in areas dominated by slumping organic soils (i.e. peat) showed elevated concentrations of Hg and toxic methylmercury (MeHg) in these systems, strongly related to inputs of organic matter and nutrients into surface waters (MacMillan et al., 2015).

Permafrost has historically been considered an impermeable barrier to the movement of contaminants due to their low infiltration and mobility in frozen soils (Grannas et al., 2013). However, ongoing and future climate warming will likely disrupt the sequestration of contaminants in permafrost soils and lead to enhanced leaching and therefore the mobility of organic and inorganic contaminants to nearby aquatic systems across all permafrost zones. This, however, will not necessarily result in increased contaminant concentrations due to a dilution effect by other materials transported with the contaminant.

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situated within the larger footprint to successfully interpret eddy flux dynamics

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These recently developed instruments that require minimal maintenance and are ideal for usage in have recently been developed, offering new opportunities for quasi-continuous gas flux measurements in remote locations. Furthermore,

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Given a sufficient power supply on-site, robust, field deployable closed-path gas analyzers are also suitable for continuous eddy flux measurements (e.g. Vesala et al., 2006).

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or CarboEurope (Papale et al., 2006),