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REVIEW

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A review of the species, community, and ecosystem impacts of road salt salinisation in fresh waters

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Abstract

1. Freshwater ecosystems worldwide are threatened by salinisation caused by human activities. Scientific attention on the ecological impacts of salinisation from road deicing salts is increasing exponentially.
2. Spanning multiple trophic levels and ecosystem types, we review and synthesise the ecological impacts of road salt in freshwater ecosystems to understand species-, community-, and ecosystem-level responses. In our review, we identify knowledge gaps that we hope will motivate future research directions.
3. We found that road salts negatively affect species at all trophic levels, from biofilms to fish. The concentration at which road salt triggered an effect varied considerably. Species-level impacts were generally sub-lethal, leading to reductions in growth and reproduction, which can be magnified by natural stressors such as predation. Community-level impacts including reductions of biodiversity were common, leading to communities of salt-tolerant species, which may have implications for disease transmission from enhanced recruitment of salt-tolerant host species such as mosquitoes. At the ecosystem level, road salts alter nutrient and energy flow. Contaminated wetlands could see greater export of greenhouse gases, streams will probably export more nitrogen and carbon, and lakes will encounter altered hydrology and oxygen dynamics, leading to greater phosphorus release from sediments.
4. While it is necessary to keep roads safe for humans, the costs to freshwater ecosystems may be severe if actions are not taken to mitigate road salt salinisation. Cooperation among policy makers, environmental managers, transportation professionals, scientists, and the public will be crucial to prevent a loss of ecosystem services including water clarity, drinkable water, recreation venues, and fisheries.

KEYWORDS
biodiversity conservation, de-icing, ecotoxicology, salinization, salinity

1 | INTRODUCTION

The salinisation of freshwater ecosystems from human activities is a global issue that threatens freshwater ecosystems and resources (Cañedo-Argüelles et al., 2013; Herbert et al., 2015). All types of freshwater ecosystems—lakes, rivers, streams, and wetlands—are affected by human-induced salinisation. Salinisation occurs from a variety of sources, including vegetation clearance, irrigation and agricultural practices, sea level rise, regulation of river hydrology, mining, and the application of road deicing salts (Albecker & McCoy, 2017; Castillo, Sharpe, Ghalambor, & De León, 2017; Herbert et al., 2015). Among these five sources, the rising salinity of fresh waters from the use of road deicing salts has received increasing attention from scientists, policy makers, environmental managers, transportation professionals, and the public (Schuler et al., 2019). Attention to road salt salinisation is growing because of the high concentrations observed in the environment, lasting ecological effects, and contamination of our drinking water (Fay & Shi, 2012; Findlay & Kelly, 2011; Tiwari & Rachlin, 2018). Even some of the largest sources of fresh water around the world have been affected. For example, in Lake Constance, Europe's second largest freshwater lake by volume, salt concentrations have more than doubled and road salts were responsible for 52% of the elevated salinity (Mueller & Gaechter, 2012). Given USD$trillion-dollar valuations of our fresh waters (Campbell, Cooper, Friedman, & Anderson, 2015), it is imperative that we understand how road salt salinisation will affect the ecology and functioning of freshwater ecosystems.

Road deicing salts are applied to increase human safety for those traveling during dangerous winter conditions. Road salts reduce accident rates on average 87% and 78% on two-lane and multilane highways, respectively (Kuemmel & Hanbali, 1992; Mullaney, Lorenz, & Arntson, 2009; Usman, Fu, & Miranda-Moreno, 2010). The most common deicers are inorganic chloride-based salts—i.e. sodium chloride (NaCl), magnesium chloride ($MgCl_2$), and calcium chloride ($CaCl_2$). After application, road salts dissolve and enter freshwater ecosystems through saline overland flow generated by snowmelt or rain and through groundwater sources (Evans & Frick, 2001). Although these inorganic salts dissolve into their constitutive ions, the chloride (Cl^-) ion is not readily bio-transformed or broken down in a way that reduces Cl^- toxicity over time. Thus, road salt is a persistent freshwater contaminant and freshwater communities can be exposed for long periods of time depending on the dynamics of surface and groundwaters (Findlay & Kelly, 2011; Ledford, Lautz, & Stella, 2016; Mason, Norton, Fernandez, & Katz, 1999). Moreover, exposure to road salt also occurs during the warm seasons because of contaminated ground waters, which are discharged into freshwater ecosystems when replaced by rainfall (Kelly et al., 2008; Todd & Kaltenecker, 2012). Road salts such as $CaCl_2$ and $MgCl_2$ are also used for dust binding on gravel roads, increasing the potential for salt contamination during warmer months (Transportation Research Board, 2007). Removal of inorganic salts from fresh waters is generally cost prohibitive, so understanding the ecological impacts and the consequences for ecosystems services valued by humans is paramount toward reaching a balance between human safety and the impacts of road salt salinisation.

Road salt contamination varies widely among freshwater ecosystems. Typical Cl^- concentrations from natural sources in surface freshwater ecosystems are <20 mg/L (Table 1). Among surface water systems, Cl^- contamination from road salts can range from 6 to 13,500 mg/L in ponds next to salt-storage facilities (Ohno, 1990). Urban streams and wetlands exhibit the highest road salt contamination among surface water systems, but urban lakes can also be highly contaminated leading to severe alterations to hydrology such as cultural meromixis—the lack of seasonal turnover due to anthropogenic sources (Bridgeman et al., 2000; Sibert, Koretsky, & Wyman, 2015). Although most freshwater ecosystems have Cl^- concentrations below the Canadian chronic threshold of 160 mg/L (Environment Canada, 2011) and the U.S. Environmental Protection Agencies (EPA) chronic threshold of 230 mg/L (Evans & Frick, 2001; USEPA, 1988), we show in this review that ecological impacts can occur below these thresholds. Further, there is little evidence that road salt pollution in freshwater ecosystems in colder regions is waning, rather concentrations continue to rise (Dugan et al., 2017; Kaushal et al., 2018).

Despite road salts affecting large regions of the world and decades of research on the subject, there have been few comprehensive review and synthesis papers on the impacts of road salt across trophic levels identifying species-, community-, and ecosystem-level

TABLE 1 A typical range of chloride concentrations (mg Cl^-/L) among inland freshwater ecosystems from natural sources and, while most contaminated systems are <200 mg Cl^-/L, a range of chloride concentrations (low to extremely high) observed resulting from road salt contamination

Freshwater ecosystem	Range due to natural sources (mg Cl^-/L)	Range of road salt contamination (mg Cl^-/L)	References
Lakes	0–10	6–1,000	Bridgeman et al., 2000; Cherkauer & Ostenso, 1976; Dixit, Dixit, Smol, Hughes, & Paulsen, 2000; Dugan et al., 2017; Evans & Frick, 2001; Judd et al., 2005; Kelting, Laxson, & Yerger, 2012; Likens & Buso, 2010; MacLeod, Sibert, Snyder, & Koretsky, 2011; Mueller & Gaechter, 2012; Novotny, Murphy, & Stefan, 2008; Sibert et al., 2015
River/stream	1–20	10–7,730	Corsi, De Cicco, Lutz, & Hirsch, 2015; Corsi et al., 2010; Crowther & Hynes, 1977; Evans & Frick, 2001; Findlay & Kelly, 2011; Hubbart, Kellner, Hooper, & Zeiger, 2017; Kaushal et al., 2005; Kelly et al., 2008; Kilgour, Gharabaghi, & Perera, 2014; Wallace & Biastoch, 2016
Wetland/pond	0–12	10–13,500	Collins & Russell, 2009; Evans & Frick, 2001; Fay & Shi, 2012; Hill & Sadowski, 2016; Ohno, 1990; Richburg et al., 2001; Sriyaraj & Shutes, 2001; Van Meter et al., 2011

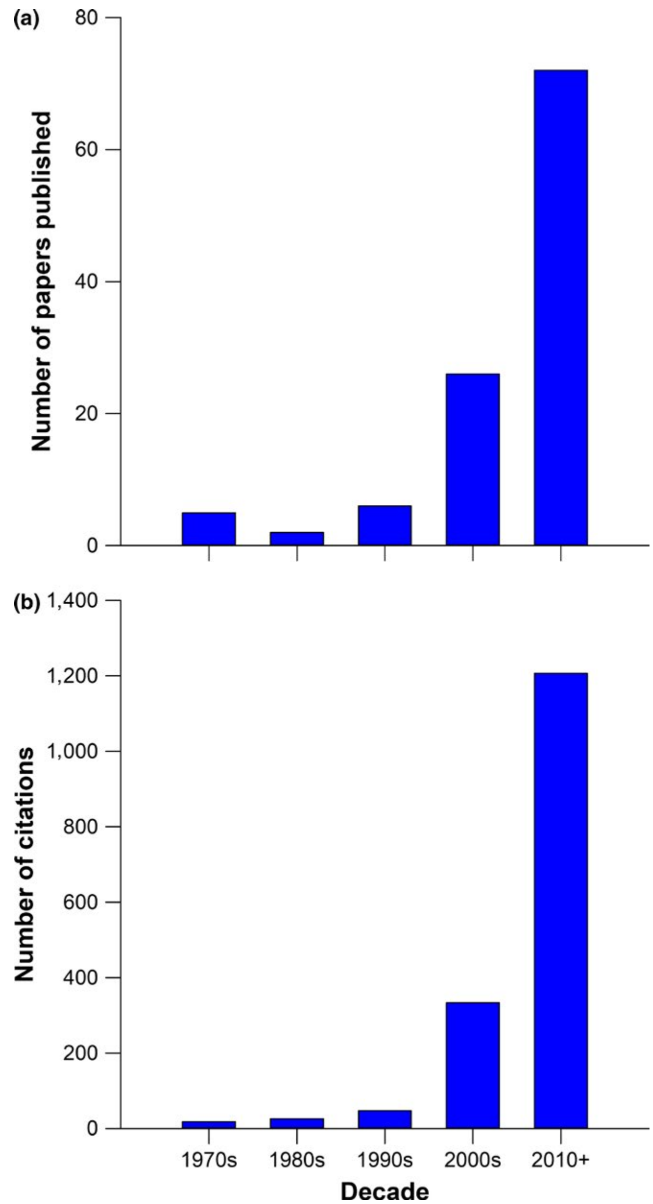

FIGURE 1 Total number of publications (a) and citations (b) from 1970 to 2017 generated from a search of Thomson Reuters' Web of Science using the search term *road salt*, *deicing*, and *deicing chemical* alone and in combination with the following search terms as *topics* among all databases: *amphibian*, *algae*, *aquatic insect*, *biogeochemistry*, *ecosystem function*, *fish*, *macroinvertebrate*, *microcrustacean*, *mollusk*, *phytoplankton*, and *zooplankton*. The focus of the search was to find studies on the ecological impacts of road deicing salts and does not include papers documenting other sources of freshwater salinisation (e.g. mining) or those that solely report on road salt concentrations in freshwater ecosystems [Colour figure can be viewed at wileyonlinelibrary.com]

impacts. Evans and Frick (2001) were the first to provide a robust assessment on the impacts of road salts on a variety of freshwater organisms. Since publication of the Evans & Frick report, the study of the ecological impacts of roads salts has increased substantially. Using the search terms *road salt*, *deicing* (or de-icing), and *deicing chemical* alone and in combination with the following search terms as topics: *amphibian*, *algae*, *aquatic insect*, *biogeochemistry*, *ecosystem function*, *fish*, *macroinvertebrate*, *microcrustacean*, *mollusk*, *phytoplankton*, and *zooplankton*, we found that among all Web of Science© databases from 1945 to 2017 an exponential increase in the number of papers examining the ecological impacts of road salts (Figure 1). Recently, Tiwari and Rachlin (2018) provided an overview of road salt concentrations in freshwater ecosystems and on the mechanisms of how road salts affect a variety of freshwater organisms. In addition to Evans and Frick (2001) and Tiwari and Rachlin (2018), we need a robust assessment not only of species-level impacts of road salts among trophic levels, but community- and ecosystem-level impacts among ecosystem types. Such a comprehensive review is important not only for future scientific study, but also to inform current environmental policy, the public, transportation professionals, and environmental managers.

Our objective was to review and synthesise the current state of knowledge regarding the ecological impacts of road salts in a way that builds on previous reviews. Specifically, we review the ecological impacts of road salt salinisation in freshwater ecosystems to understand species-, community-, and ecosystem-level responses spanning multiple trophic levels and ecosystem types. In our review, we identify knowledge gaps to motivate future research directions. Given that the most commonly used road deicer by far is NaCl (Evans & Frick, 2001) and the paucity of information on the ecological effects of alternatives such as $MgCl_2$ and $CaCl_2$ (Schuler et al., 2017), the impacts of NaCl is the primary focus of our review. Chloride is generally one of the most definitive indicators of road salt pollution (Schuler et al., 2019). Some field or laboratory studies on road salt report conductivity, rather than Cl^- concentration. Several ecosystem-specific water quality characteristics such as calcium concentration can influence conductivity, making it difficult to assess what contribution road salt might have to the conductivity measurements. Therefore, unless the Cl^- concentration, the NaCl concentration, or the system-specific chloride-conductivity equation was reported in the original study, we did not convert conductivity to mg Cl^-/L from the cited literature.

2 | SPECIES-LEVEL RESPONSES

2.1 | Basal trophic levels

Basal trophic levels (i.e. producers and decomposers) are important for transforming and transferring energy to higher trophic levels and maintaining ecosystem functions such as nutrient cycling and decomposition (Covich, Palmer, & Crowl, 1999; Wetzel, 1995). Bacteria, phytoplankton, periphyton, macroalgae, and plants also provide important habitat and food resources for other freshwater organisms. Few studies have investigated the effects of road salt salinisation on basal trophic levels. In a microcosm study of stream microbial communities using NaCl, Cochero, Licursi, and Gomez (2017) found bacterial densities were diminished at conductivities around 1,500 μS/cm during a 72-hr exposure. The NaCl treatment was discontinued and after 144 hr, bacterial densities returned to similar densities as uncontaminated control conditions. This indicates that bacteria may recover quickly after a short-term exposure to elevated NaCl concentrations. Some species of bacteria are tolerant to elevated NaCl concentrations (Dickmann & Gochnauer, 1978). In one instance, road salt concentrations of 2,000–5,000 mg Cl^-/L increased the diversity of denitrifying bacteria species composition in roadside wetlands compared to forested wetlands unexposed to road salts (Lancaster, Bushey, Tobias, Song, & Vadas, 2016). At present, we do not understand how microbial species composition and turnover is affected by lower, but more environmentally common road salt concentrations. This is critically important because if bacterial function is reduced due to changes in species composition, trophic transfer of energy and ecosystem functions may be altered. Although bacterial densities can recover quickly (Cochero et al., 2017), most freshwater systems exhibit sustained, elevated concentrations of road salt. Thus, it is probable that changes in the number and type of bacterial species have occurred over several decades. The consequences of changes in bacterial communities and legacy effects on microbial communities should be a major focus of future research.

Among basal trophic levels, photosynthetic organisms have variable responses to road salt contamination. Diatoms exhibit significant species turnover with elevated salt concentrations in streams (Porter-Goff, Frost, & Xenopoulos, 2013) and lakes (MacDougall et al., 2017). For example, a reduction in several diatom species (e.g. *Diatoma vulgaris*, *Encyonema caespitosum*, *Pinnularia microstauron*) occurs in streams with salt concentrations ≥35 mg Cl^-/L (Porter-Goff et al., 2013), a low threshold considering that road salt concentrations in streams can commonly exceed 5,000 mg Cl^-/L (Kaushal et al., 2005). In semi-natural experimental freshwater communities, the biomass of filamentous green algae (i.e. charophytes) was reduced by 55–99% at concentrations of 250–1,000 mg Cl^-/L (Hintz et al., 2017). Another study showed 1,000 mg Cl^-/L reduced chlorophyll *a* content and biomass of the macroalgae *Nitella* sp. (Lind et al., 2018). Some evidence suggests that road salt can have a stimulating effect on phytoplankton abundance (Fay & Shi, 2012; Hintz et al., 2017). In Third Sister Lake (Michigan, U.S.A.), Judd et al. (2005) suggested phytoplankton species from the genus *Asterionella* that were associated with brackish environments were common in Third Sister Lake at concentrations of 217–445 mg Cl^-/L. However, *Asterionella* also appear to be more common in lakes experiencing cultural eutrophication (Spaulding & Edlund, 2009). This highlights an important consideration in the study of road salts; many systems experiencing road salt salinisation are simultaneously experiencing eutrophication (Judd et al., 2005). Thus, in natural systems, it is difficult to identify the separate impacts of road salt from other factors that may also be affecting the system or ecological community.

Our understanding of the impacts of road salts on phytoplankton and photosynthetic periphyton species is deficient. Moreover, we do not understand how changes in species composition will affect higher trophic levels. For example, changes in the occurrence of various diatoms or phytoplankton species may correspond to stoichiometric changes of algal food resources. Algal stoichiometry is an important factor regulating zooplankton growth rates, reproductive traits, and population growth (Lind & Jeyasingh, 2017). Thus, future research on patterns of change in phytoplankton and photosynthetic periphyton species and the implications of such change is needed.

Among basal trophic levels, macrophytes are important drivers of animal species richness and ecosystem function (Carpenter & Lodge, 1986; Engelhardt & Ritchie, 2001; Thomaz & Cunha, 2010). Very little is known about the impacts of road salt on freshwater plants. Concentrations of 300–1,500 mg Cl^-/L reduced the growth of a peat moss (*Sphagnum recurvum*) by 17–57% and may cause the elimination mosses from highly contaminated bogs (Wilcox, 1984). In a bog in Indiana (U.S.A.), one study noted that all endemic plant species (except some moat species) were absent from the bog when salt concentrations reached 1,215 mg Cl^-/L; however, after a 50% reduction in road salt concentrations, many of the endemic bog plants returned (Wilcox, 1986). Some freshwater species are more tolerant of high NaCl concentrations. For example, the ubiquitous sago pondweed (*Potamogeton pectinatus* or *Stuckenia pectinata*) starts growing tubers (a stress response) when salt concentrations reached about 1,800 mg Cl^-/L (Teeter, 1965). Sago growth was completely inhibited at concentrations of 5,500 mg Cl^-/L in 1-week-old plants, but it required 9,000 mg Cl^-/L to inhibit growth of plants that were 4–8 weeks old; the latter concentration was often fatal (Teeter, 1965).

We clearly need much more research on obligate freshwater macrophytes before we can arrive at robust generalities. We might also expect differences in sensitivity between emergent versus submerged macrophytes, given that the former would have a portion of its tissues unexposed to salty water. Long-term macrophyte surveys linked with measures of road salt pollution (i.e. Cl^- concentration, conductivity, salinity) could yield important correlative insights on the loss of macrophyte species and biomass with increasing salt concentration. Experimental work at multiple spatial scales (e.g. mesocosm or in situ experiments) is also needed to identify mechanisms behind road salt toxicity on freshwater macrophytes. At this point, we lack a keen understanding of how road salts affect common freshwater macrophytes that promote animal species richness and generate important ecosystem functions.

2.2 | Zooplankton

Zooplankton play a critical functional role in freshwater systems as grazers, predators, and prey. They transfer energy from primary producers into biomass that creates food resources for higher trophic level consumers (e.g. fish) and regulate the cycling of water clarity, energy, and nutrients (Carpenter, Kitchell, & Hodgson, 1985; Pace & Orcutt, 1981). Among the three major groups of zooplankton—cladocerans, copepods, and rotifers—the effects of road salts have primarily focused on a single family of cladocerans, the Daphniidae. We lack information on the impacts of road salt on the largest predatory cladocerans such as *Polyphemus* and *Bythotrephes*. Sarma, Nandini, Morales-Ventura, Delgado-Martinez, and Gonzalez-Valverde (2006) demonstrated that five species of cladocerans (*Alona rectangula*, *Ceriodaphnia dubia*, *Daphnia pulex*, *Moina macrocopa*, and *Simocephalus vetulus*) were unable to survive and reproduce at levels of 3,000 mg Cl^-/L, with reduced survival and reproduction occurring at 910 mg Cl^-/L. For *Daphnia magna*, lethal concentrations at which 50% of the population dies (i.e. LC_{50}) generally exceed 2,800 mg Cl^-/L (Goncalves, Castro, Pardal, & Goncalves, 2007; Martinez-Jeronimo & Martinez-Jeronimo, 2007; Mount, Gulley, Hockett, Garrison, & Evans, 1997). Over a period of 21 days, the LC_{50} for *Daphnia longispina* was 1,334 mg Cl^-/L (Goncalves et al., 2007). Ubiquitous among freshwater ecosystems, *Ceriodaphnia dubia* exhibits an IC_{25} value—the *inhibition concentration* that causes a 25% reduction of growth or reproduction—at 1,050 mg Cl^-/L (Corsi, Graczyk, Geis, Booth, & Richards, 2010). Road salt concentrations of 860 and 1,300 mg Cl^-/L can reduce *Daphnia pulex* abundance by 40 and 79%, respectively (Hintz & Relyea, 2017b). Searle, Shaw, Hunsberger, Prado, and Duffy (2016) found that experimental *Daphnia dentifera* population densities were reduced by almost 50% at 364 mg Cl^-/L, which is similar to the effects observed on large-bodied grazer such as *Daphnia* in a natural lake contaminated with 217–445 mg Cl^-/L (Judd et al., 2005).

Laboratory tests that quantify LC_{50} values are valuable, but it is likely that even sublethal concentrations can affect zooplankton population abundance. Searle et al. (2016) demonstrated the contrast between population- versus individual-level impacts and the importance of sub-lethal effects. Cladocerans may be tolerant of road salt concentrations currently observed in lake ecosystems (Dugan et al., 2017), but the sublethal effects on life history characteristics (age at first reproduction, brood number, brood size) and the toxic effects on neonates may prevent population growth (Goncalves et al., 2007). Given that most large-bodied cladocerans are important phytoplankton grazers that strongly regulate water clarity and are an important food resource for fish, a loss of large-bodied cladocerans would be expected to trigger increases in phytoplankton and losses of water clarity and fish recruitment (discussed further in *Community-level responses* below).

Copepods are the most diverse group of zooplankton, but we know very little regarding the species-level impacts. A search of Web of Science using *copepod* and *road salt* yielded five papers, all of which examine the effects on road salt in a community context (discussed in the *Community-level responses* section below). Evans and Frick (2001) report one instance where the cyclopoid copepod species *Cyclops serrulatus* tolerated a maximum NaCl concentrations of 239 mg Cl^-/L and *Cyclops vernalis* and *Diaptomus oregonensis* were apparently immobilised at concentrations of 3,687 and 1,843 mg Cl^-/L, respectively. More research on the effects of road salt on copepods is needed because copepods, like cladocerans, are important food resources for other zooplankton and fish and the top-down control of algae in freshwater systems (Sommer & Sommer, 2006).

Rotifers consume protists, bacteria, and algae; like other zooplankton, they are important for energy transfer to higher trophic levels. As with copepods, we know very little about the species-level effects of road salt on rotifers. Sarma et al. (2006) tested the effects of NaCl on five rotifer species: *Anuraeopsis fissa*, *Brachionus calyciflorus*, *Brachionus havanaensis*, *Brachionus patulus*, and *Brachionus rubens*. Concentrations of 910 mg Cl^-/L reduced the abundance of *A. fissa*, *B. calyciflorus*, and *B. havanaensis*, but the abundances of *B. patulus* and *B. rubens* were not reduced until concentrations reached 1,820 mg Cl^-/L. This two-fold difference in the concentration among the species suggests that rotifer species may possess a wide range of tolerance to road salts. Much more research is necessary to understand the responses of rotifers to road salt contamination.

2.3 | Macroinvertebrates

Freshwater macroinvertebrates perform a variety of functions through the processing of multiple types of autochthonous or allochthonous organic matter (Covich et al., 1999). Macroinvertebrates are also critical food resources for higher trophic levels in freshwater food webs. Road salt contamination can produce a dense saline layer just above the sediment–water interface, which may expose macroinvertebrates to much higher salt concentrations than indicated by pelagic water monitoring surveys (Ellis, Champlin, & Stefan, 1997; Eyles & Meriano, 2010; Gillis, 2011; Novotny & Stefan, 2012b). In this sub-section, we review single-species studies as well as mesocosm studies in which direct toxicity of road salt can be inferred.

Freshwater insects can be relatively tolerant to high road salt concentrations but there are some substantial differences among reported impacts. Sub-lethal impacts include increased drift behaviour by stream insects when salt concentrations exceed 1,000 mg Cl^-/L (Crowther & Hynes, 1977), but mortality does not appear to occur until uncommonly high concentrations of road salt. In a laboratory study, Blasius and Merritt (2002) observed no mortality at concentrations up to 6,066 mg Cl^-/L on two perlid species of stonefly (Plecoptera: *Acroneuria abnormis*, *Agnetina capitata*) and one species of tipulid crane fly (Diptera: *Tipula abdominalis*) after a 96-hr exposure. The authors concluded that short-term (96-hr) acute exposures to 6,066 mg Cl^-/L of road salt did not pose a risk to macroinvertebrates in Michigan (U.S.A.) streams contaminated with road salt. Tolerance to high concentrations were also reported by Benbow and Merritt (2004), who found that the 96-hr LC_{50} of *Callibaetis fluctuans* (Ephemeroptera) exceeded 2,558 mg Cl^-/L and that of the common glassworm *Chaoborus americanus* (Diptera) exceeded 4,502 mg Cl^-/L. A recent stream mesocosm study by Clements and Kotalik (2016) showed that insects in non-polluted headwater streams were highly vulnerable to changes in stream specific conductance from ions of various salts, including those found in road salt. They concluded that the USEPA's chronic threshold of 230 mg Cl^-/L may not protect many insect taxa because laboratory strains are often more tolerant than organisms collected from a natural system. It may be that induced or evolved tolerance to road salt may be occurring on a landscape scale. It is possible that prior exposure to road salts are indicating that many species of insects are more tolerant to road salts compared to previously unexposed strains from natural systems (Clements & Kotalik, 2016; Kotalik, Clements, & Cadmus, 2017). As such, toxicity tests from laboratory studies that use insects previously exposed to high road salt concentrations over multiple generations may underestimate the sensitivity of naïve individuals in uncontaminated systems (also see Brady, Richardson, & Kunz, 2017).

There are multiple areas in need of further research regarding freshwater insects. Although short-term toxicity tests have yielded insights into the effects of road salt on mortality and drift behaviour, we do not know how road salt affects emergence dynamics (e.g. timing). Experiments following the aquatic and terrestrial life stages of insects would yield important insights into effects that may transfer to the terrestrial stages, which may generate a feedback loop to the aquatic stages. It is also likely that road salt has sub-lethal and long-term impacts on reproduction and recruitment (Benbow & Merritt, 2004; Blasius & Merritt, 2002). Experiments are needed to assess whether road salt affects behaviours such as oviposition, which may have implications for aquatic–terrestrial food web linkages.

Road salt alters the behaviour, abundance, and mortality of freshwater amphipods, which are important diet items for many economically important fish species (Liao, Pierce, & Larscheid, 2002; Tyson & Knight, 2001). Concentrations of 606, 1,516, and 6,066 mg Cl^-/L increases drift behaviour by 13%, 45%, and 58% by *Gammarus pseudolimnaeus*, respectively (Blasius & Merritt, 2002). In semi-natural mesocosm experiments, concentrations <727 mg Cl^-/L do not appear to affect abundance of the sentinel species *Hyalella azteca* (Stoler, Walker et al., 2017), but 1,000 mg Cl^-/L caused a 92% reduction of this species in a separate mesocosm experiment (Hintz et al., 2017). In laboratory studies, mortality of *H. azteca* was below 50% at 4,502 mg Cl^-/L (Benbow & Merritt, 2004) and the 96-hr LC_{50} of *Gammarus pseudolimnaeus* was 7,700 mg Cl^-/L (Blasius & Merritt, 2002). Mortality of adult amphipods appears to occur at high concentrations, so it is possible the significant reduction in amphipods from Hintz et al. (2017) resulted from sub-lethal effects either on reproduction or toxicity to the offspring, but this hypothesis needs to be addressed experimentally.

Freshwater gastropods strongly regulate periphytic and epiphytic production in freshwater systems and they exhibit positive and negative responses to road salt. Concentrations up to 1,000 mg Cl^-/L increase growth, reproduction, and abundance in a ubiquitous pond snail (*Physa acuta*; Hintz et al., 2017; Kefford & Nugegoda, 2005). Suski, Salice, and Patino (2012) showed that two ubiquitous snail species differed in their susceptibility to NaCl. Ram's horn snails (*Helisoma trivolvis*) did not reproduce when conductivity was ≥3,000 μS/cm (of NaCl) and they exhibited a 30% reduction in survival at 4,000 μS/cm. At the same concentrations, survival and reproduction of the glossy Physa (*Physa pomilia*) was unaffected. In a mesocosm experiment, the banded mystery snail (*Viviparus georgianus*) experienced 5–11% mortality in road salt concentrations ≥500 mg Cl^-/L after a week-long exposure (Hintz et al., 2017). As is the case with other macroinvertebrates, we only understand the effects of road salts on a few species of gastropods. A broader understanding of the direct effects of road salts on freshwater gastropods is needed, particularly given that some species respond positively while other species respond negatively.

Unionid mussels are important in filtering phytoplankton from the water column. They are declining globally, but it is unknown if road salts are contributing to these declines. Short-term spikes in road salt concentrations ≥113 mg Cl^-/L can reduce the viability of glochidia (i.e. larvae) and their successful attachment to fish, which may reduce successful reproduction of freshwater mussels (Beggel & Geist, 2015; Gillis, 2011; Nogueira et al., 2015). Road salt can also increase the filtration behaviour of freshwater mussels, suggesting a need to flush road salt from the body (Hartmann, Beggel, Auerswald, Stoeckle, & Geist, 2016). Many juvenile mussels remain burrowed during the early-life stages. As a result, the contamination of hyporheic zones and ground waters (not just surface waters) may also cause negative impacts on freshwater mussels. In fact, up to 20% mussel mortality can occur in stream sediments contaminated with 3,000–4,000 mg Cl^-/L (Roy, McInnis, Bickerton, & Gillis, 2015). It is probable that road salts are inhibiting mussel recovery or triggering declines (Prosser, Rochfort, McInnis, Exall, & Gillis, 2017; Todd & Kaltenecker, 2012), but this area of research needs more scientific attention.

2.4 | Amphibians

In freshwater systems, amphibians regulate primary production and community structure (Hocking & Babbitt, 2014). While some species live in lakes and streams, a large fraction of species live in wetlands or ponds that can experience large inputs of salt, particularly in seasonal stormwater or treatment ponds (Van Meter, Swan, & Snodgrass, 2011). Amphibians are declining globally at rates much faster than other vertebrate groups (IUCN, 2017). Auditory and visual field surveys of eggs, larvae, and adults suggest that spotted salamanders (*Ambystoma maculatum*) and wood frogs (*Lithobates sylvaticus*) are rarely found in ponds containing >200 mg Cl^-/L, but spring peepers (*Pseudacris crucifer*) and green frogs (*Lithobates clamitans*) were comparatively more tolerant to salt (Collins & Russell, 2009). The embryos and tadpoles of American toad (*Anaxyrus americanus*) and American bullfrog (*Lithobates catesbeianus*) appear to be unaffected by road salt concentrations <1,000 mg Cl^-/L (Collins & Russell, 2009; Matlaga, Phillips, & Soucek, 2014).

Road salt can have dramatic effects on the embryonic stages of some amphibians. In natural wetlands, spotted salamander (*Ambystoma maculatum*) embryos are much smaller near highways than those far from highways (Karraker & Gibbs, 2011a). Embryonic survival of the spotted salamander is reduced to 68% and 3% in moderate (145 mg Cl^-/L) and high (945 mg Cl^-/L) concentrations, respectively (Karraker, Gibbs, & Vonesh, 2008). Embryonic survival of wood frog embryos is reduced from 91% to 41% in road salt concentrations of 945 mg Cl^-/L (Karraker et al., 2008). At the larval stage, road salt concentrations reduced spotted salamander (*Ambystoma maculatum*) mass by 18% at 145 mg Cl^-/L and 33% at 945 mg Cl^-/L, perhaps due to a reduced capacity to osmoregulate (Karraker & Gibbs, 2011b). Interestingly, mass loss at 145 mg Cl^-/L was almost completely regained when salamanders were subsequently returned to low concentrations (1 mg Cl^-/L). However, mass loss continued when the salamanders were initially exposed to 945 mg Cl^-/L and then returned to low concentrations (Karraker & Gibbs, 2011b). In the same concentration, survival of larval wood frogs decreased from 64% to 20% (Karraker et al., 2008).

Recent research shows that sublethal salt concentrations can masculinise wood frog metamorphs (Lambert, Stoler, Smylie, Relyea, & Skelly, 2017). Lambert et al. (2017) found that road salt concentration of 867 mg Cl^{-}/L decreased the ratio of females in an experimental population by 10%. More research is needed on the role road salt plays in determining the sex of amphibians—the ongoing decline of amphibian diversity and abundance might potentially be exacerbated by such sublethal effects. Importantly, sex change in frogs might suggest the possibility that salt may cause sex changes in other aquatic species (not just amphibians), which remains to be studied.

At present, we have a limited understanding about the sensitivity of amphibian embryos and larvae to road salts, but we know almost nothing about the sensitivity of adults in terms of their survival, growth, and reproduction. Future research should examine the impacts of salt on adult frogs and salamanders and how contaminated habitats affect oviposition. Several studies have shown that amphibians can induce a tolerance to widespread contaminants such as pesticides (Hua, Jones, & Relyea, 2014; Hua et al., 2015). It remains to be explored whether amphibians can evolve or induce a tolerance to road salt, which may reduce harmful impacts. It is also unknown whether road salt can disrupt the sensory abilities of amphibians, potentially exposing embryos and offspring to predators.

2.5 | Fish

Fish have important cascading effects in freshwater ecosystems that can regulate primary and secondary production (Carpenter et al., 1985). They also generate valuable ecosystem services through commercial and recreational fisheries. Few studies have examined the effects of environmentally relevant concentrations of road salts on fish. In Atlantic salmon (*Salmo salar*), high road salt concentrations (3,000 mg Cl^{-}/L) reduce egg swelling and egg survival (Mahrosh et al., 2014). At similar concentrations, Tollefsen et al. (2015) found that road salt interferes with osmoregulation, oxidative stress, metabolism, renal function, and development of Atlantic salmon embryos. Reduced weight and survival of the ubiquitous fathead minnow (*Pimephales promelas*) occurred at salt concentrations of 2,920 mg Cl^{-}/L (Corsi et al., 2010). Mosquitofish can tolerate salinity levels in excess of 15,200 mg Cl^{-}/L (Nordlie & Mirandi, 1996) and exhibit no adverse effects of elevated salinity, even in environments where they are considered invasive (Karraker, Arrigoni, & Dudgeon, 2010). In a semi-natural experimental community, salt increased the mass of bridle shiners (*Notropis bifrenatus*) at concentrations of 500–1,000 mg Cl^{-}/L (Hintz et al., 2017). Newly hatched rainbow trout alevins (*Oncorhynchus mykiss*) exposed to NaCl road salt for 25 days (well after they developed into free-swimming fry) experienced a 9% reduction in length and 27% reduction in mass at 3,000 mg Cl^{-}/L for NaCl (Hintz & Relyea, 2017a). In a subsequent experiment, the same species was tested at a later life stage and NaCl road salt at 2,000 and 4,000 mg Cl^{-}/L did not affect rainbow trout growth (Hintz & Relyea, 2017b). Compared with other freshwater species, fish appear to be relatively tolerant to salt, but a broader assessment of obligate freshwater fish species is needed because reductions in fish growth may influence recruitment and population growth (Milner et al., 2003).

As is evident from the above studies, life stage and species identity are important factors determining susceptibility to road salt, but too few fish species have been studied to arrive at many generalities. We clearly need many more studies of fish susceptibility, with a focus on threatened and endangered freshwater fish species whose recovery may be impeded by salt pollution. The early-life stages of fish appear to be the most susceptible to deicing salts, perhaps because they lack fully developed organs associated with osmoregulation (e.g. kidneys and gills), but the mechanisms remain unclear. As we expand our knowledge on how road salt affects freshwater fish and other organisms at multiple trophic levels, it would be valuable to develop population growth models in response to salt sensitivity as well as food-web models to forecast changes in energy flow through food webs due to road salt salinisation.

3 | COMMUNITY-LEVEL RESPONSES

As reviewed in the previous section, road salt affects many species across all trophic levels at environmentally relevant concentrations. Such direct lethal and sublethal impacts should lead to community-level responses through indirect, interactive (i.e. synergistic or antagonistic), and additive effects. In this section, we review community-level responses by ecosystem type: lakes, streams and rivers, and wetlands because community composition and interactions are often unique in each ecosystem.

3.1 | Lake communities

Compared to rivers, streams, and wetlands, few studies have examined the impacts of road salts on lake communities. In a natural lake, road salt concentrations of 217–445 mg Cl^{-}/L have been associated with changes in the species composition of phytoplankton, zooplankton, and benthic communities towards more salt-tolerant species (Bridgeman et al., 2000; Judd et al., 2005). From the same lake, the authors suggested that sediment cores revealed that following road salt contamination, one genus of phytoplankton common to mesotrophic and brackish waters (*Asterionella*) became far more prevalent while large-bodied zooplankton grazers like *Daphnia* declined; as a result, there was reduced grazing pressure on phytoplankton communities (Judd et al., 2005), which may lead to elevated phytoplankton abundance.

Predators can have strong consumptive and non-consumptive effects on prey (e.g. Bourdeau, Bach, & Peacor, 2016; Carpenter et al., 1985) and recent studies show that predator effects can interact with road salt. One study found that predatory fish acted synergistically to increase the mortality of native clams (*Sphaerium simile*) at 1,000 mg Cl^{-}/L, perhaps due to the costs of predator avoidance and osmotic stress (Hintz et al., 2017). Hintz et al. (2017) also showed that concentrations of 1,000 mg Cl^{-}/L and fish predation had a negative synergistic effect on the richness and abundance of experimental zooplankton communities. Although this effect triggered a trophic cascade leading to phytoplankton blooms, the authors found few effects in the absence of fish predation on zooplankton communities, perhaps due to rapid adaptation of zooplankton to the salt concentrations (Coldsnow, Mattes, Hintz, & Relyea, 2017; Hintz, Jones, & Relyea, 2019). Road salt concentrations ≥860 mg Cl^-/L and non-consumptive predatory stress can also negatively affect zooplankton abundance in an additive way, and natural stress responses by zooplankton to predators (e.g. resting egg production in *Daphnia pulex*) can be inhibited by osmotic stress resulting from road salt concentrations ≥230 mg Cl^-/L (Hintz & Relyea, 2017b). In the absence of fish predation, one study found a direct negative effect on zooplankton richness and abundance at experimental concentrations of 250 Cl^-/L, highlighting the variable responses of lake zooplankton communities to road salt contamination (Sinclair & Arnott, 2018).

Water chemistry is important when considering the impacts of road salts (Elphick, Bergh, & Bailey, 2011; Mount et al., 1997, 2016). In lake ecosystems with low calcium (Ca) concentrations, road salt might lead to negative effects on zooplankton species richness and abundance at much lower Cl^- concentrations than systems with higher Ca concentrations (Jensen, Meland, Schartau, & Walseng, 2014). In two low-Ca European lakes recovering from acidification, Jensen et al. (2014) found that the recovery of the zooplankton community in the lake with a road salt concentration of 6.9 mg Cl^-/L (average Ca concentration of 0.9 mg/L) was slower than the lake with a concentration of 3.2 mg Cl^-/L (average Ca concentration of 1.1 mg/L). However, Jensen et al. (2014) found no significant indication that Cl^- concentration was solely responsible and added that the low Ca concentration in the lake with greater road salt contamination may have also been responsible for the slower recovery of the zooplankton community (also see Ashforth & Yan, 2008; Hessen, Alstad, & Skardal, 2000).

Combining our assessment of the community- and species-level effects of road salts in lake systems, we might expect dramatic shifts in diversity and food web dynamics with elevated road salt concentrations (Figure 2). Since road salt can trigger trophic cascades that cause algal blooms; shading from these phytoplankton blooms could exacerbate any direct toxic effects of road salts on benthic communities through a reduction of benthic primary production (i.e. food resources). Road salt concentrations increase with depth because of density gradients and can cause dimictic systems to become

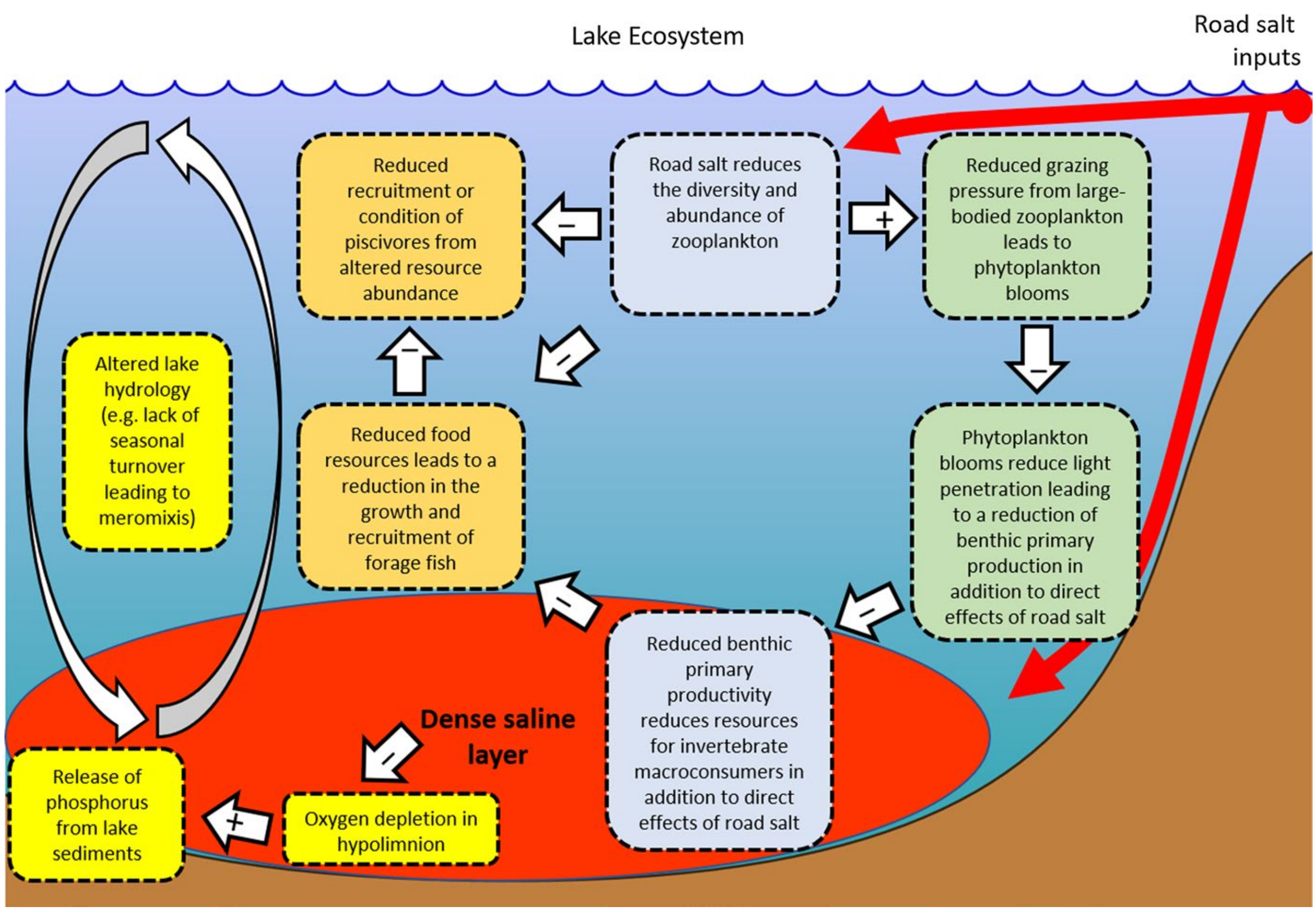

FIGURE 2 Summary of the observed and predicted species-level, community-level, and hydrological impacts on lake ecosystems. Cascading and bottom-up effects combined with reductions in the growth, abundance, and diversity of lake organisms, potentially exacerbated by salinity gradients and altered hydrologic characteristics leading to losses of ecosystem services [Colour figure can be viewed at wileyonlinelibrary.com]

meromictic around 180–241 mg Cl^-/L and trigger oxygen depletion in the hypolimnion (Bridgeman et al., 2000; Ficker et al., 2018; Jensen et al., 2014). Such oxygen depletion can release phosphorus from lake sediments (Novotny & Stefan, 2012; Wyman & Koretsky, 2018), potentially exacerbating phytoplankton blooms from the loss of large-bodied zooplankton. Thus, a dense saline layer at the bottom of lakes combined with altered lake hydrology could compound cascading effects on benthic communities. A loss of large-bodied zooplankton and a reduction in macroinvertebrate diversity might also affect higher trophic levels. Plankton and macroinvertebrates are essential diet items for most fish across multiple life stages. Changes in the abundance and diversity of these resources could have bottom-up effects that reduce fish recruitment. A reduction in the recruitment of some fish species could then negatively affect the piscivores in a lake food web. Further, a loss in water clarity due to elevated phytoplankton blooms could alter the foraging success and subsequent condition of visual predators. Ultimately, changes to lake ecosystems resulting from high road salt concentrations could be dramatic. We need more field studies, experiments, and models to better understand how road salts will change lake ecosystems and what the implications are for lake ecosystem services. Whole-lake experiments would yield important insights into the impacts of road salt salinisation in lakes. This would be valuable for controlling other factors (e.g. eutrophication) altering lake communities and food webs simultaneously.

3.2 | Stream and river communities

Despite the large number of studies documenting road salt contamination in streams, there are surprisingly few studies that examine the impacts of road salt on stream communities. Using NaCl, Cañedo-Argüelles et al. (2016) found salinity stress occurring in an invertebrate predator at 7,000 μS/cm was insufficient to trigger a trophic cascade in artificial streams. In a separate experiment, salt pulses increased the abundance of salt-tolerant invertebrate species in streams without changing species richness (Cañedo-Argüelles et al., 2014). Among basal trophic levels, an in situ experiment showed that the standing crop and diversity of autotrophs can be reduced significantly in streams at concentrations >600 mg Cl^-/L (Dickmann & Gochnauer, 1978). We might also expect synergistic effects between predatory stress and road salt on stream communities. For example, predatory stress makes sublethal pesticide concentrations 2–46 times more lethal (Relyea, 2003, 2004; Relyea & Mills, 2001) and the synergistic effects between road salts and predation can trigger trophic cascades in lake communities. However, predator cues and road salt (2,000–4,000 mg Cl^-/L) had independent effects rather than interactive effects on the growth or behaviour of stream-dwelling rainbow trout (Hintz & Relyea, 2017b).

Field surveys appear to yield dramatically different results than experimental studies of stream communities. For example, an analysis of 41 streams across a salinity gradient found that the diversity of diatoms was significantly reduced at 35 mg Cl^-/L (Porter-Goff et al., 2013). Another field study of 10 watersheds found that macroinvertebrate community composition changes occur at salt concentrations as low as 50–90 mg Cl^-/L (Wallace & Biastoch, 2016). Road salt can also alter the diversity of higher-level consumers in streams. Elevated Cl^- concentrations between 33 and 108 mg/L can reduce diversity of stream fish assemblages (Morgan et al., 2012). These field data indicate that the diversity of stream ecosystems can decrease at concentrations much lower than Cl^- concentration thresholds set by governments to protect freshwater biota.

At present, we expect reductions in the diversity among trophic levels in streams contaminated with road salt. Field studies suggest that such changes will occur at much lower concentrations than experimental studies, but it is inherently difficult to draw comparisons between the two because a host of conditions unique to an experiment or field study. Future research identifying the indirect effects of road salt in streams is needed. For instance, we do not understand how reductions in diversity or functional feeding groups in streams will alter primary production or nutrient/carbon cycling. It is possible that changes in the diversity of basal trophic levels could lead to increases or decreases in nutrient spiraling (e.g. spiral length), which would affect the exports of nutrients or energy from stream ecosystems (see *Ecosystem-level responses*). Streams exhibit severe pulses of road salt compared to lentic systems. Currently, we do not know how pulses and presses of road salt may affect community interactions, community resilience (e.g. after large pulses), and diversity in streams, which should be a focus of future research.

3.3 | Wetland communities

Road salt can also have dramatic effects in permanent and seasonal wetland communities (including stormwater and treatment ponds). At the basal level of the community, road salt contamination can reduce the richness, evenness, and coverage of native plants in natural wetlands at relatively low concentrations (i.e. 54 mg Cl^-/L; Richburg, Patterson, & Lowenstein, 2001). Tadpole grazing combined with 645 mg Cl^-/L can reduce microbial respiration by as much as 24%, which may reduce energy flow provided by detrital resources in experimental pond food webs (Van Meter, Swan, & Trossen, 2012).

As with lake communities, road salt in experimental wetland and stormwater pond communities can trigger cascading effects by killing zooplankton, which leads to elevated phytoplankton abundance due to reduced zooplankton grazing (Dananay, Krynak, Krynak, & Benard, 2015; Jones et al., 2017; Van Meter & Swan, 2014; Van Meter et al., 2011). This can occur at similar Cl^- concentrations in lakes. A decline of >50% of cladoceran zooplankton occurred in concentrations of 469 mg Cl^-/L after 2 weeks and cladocerans were almost absent from the semi-natural mesocosm communities after 7 weeks at concentrations of 246 mg Cl^-/L (Petranka & Francis, 2013). Road salt can also reduce copepod densities in experimental wetland communities >800 Cl^-/L (Stoler, Walker et al., 2017), but this negative effect depends on the quality of allochthonous resources such as leaf litter (Stoler, Hintz et al., 2017). Further, road salt interacts with abiotic conditions in wetlands. Silver, Rupprecht, and Stauffer (2009) showed that road salt (<3,000 mg Cl^-/L) can be more lethal to chironomid larvae at warmer temperatures (≥22°C). Cascading effects in wetlands suggests that zooplankton in semi-natural communities might be more susceptible to lower salt concentrations compared to many of the species-level laboratory studies. More research is needed to determine how cascading responses are modulated by abiotic conditions in wetlands (e.g. pH, dissolved oxygen, temperature).

Amphibian species richness declines with increasing Cl^- concentration in roadside ponds (Collins & Russell, 2009). Such declines may not only result from direct effects of road salt on amphibians, but indirect effects within food webs. Although larval salamander predators (*Ambystoma maculatum*) are relatively tolerant to road salts compared to zooplankton prey, because zooplankton prey communities collapsed at concentrations >246 mg Cl^-/L, an indirect reduction in salamander growth occurred (Petranka & Francis, 2013). Regarding potential interactive effects of predatory stress and road salts, Karraker et al. (2010) found no synergistic effects between a fish predator and high salt (200–4,000 mg Cl^-/L) in an experiment on five amphibian species. While higher salt concentrations may affect amphibian survival and growth at the species level, indirect effects from community interactions might reduce amphibian richness at lower concentrations.

Many wetland organisms can become more susceptible to disease when exposed to anthropogenic contaminants (Johnson et al., 2007; Taylor, Williams, & Mills, 1999). Karraker and Ruthig (2009) found no interactive effects between high salt and ubiquitous water moulds that infected amphibian clutches. Stockwell, Clulow, and Mahony (2015) actually found that NaCl concentrations >1,213 mg Cl^-/L reduced host infection loads of chytrid fungus (*Batrachochytrium dendrobatidis*) in the eastern dwarf tree frog (*Litoria fallax*). However, in a recent study, Milotic, Milotic, and Koprivnikar (2017) found that wood frogs exposed to road salt concentrations >360 mg Cl^-/L had more helminth parasite cysts and exhibited reduced anti-parasite behaviour compared to concentrations <100 mg Cl^-/L. Road salt concentrations >600 mg Cl^-/L also increase parasite infections in wood frog tadpoles (*Lithobates sylvaticus*) (Buss & Hua, 2018). It appears that the interaction between road salts and disease may largely depend on host–parasite identity.

Salinisation of wetland communities may have implications for the transmission of diseases by affecting the abundance of pathogen hosts. For example, Petranka and Doyle (2010) found that salinised habitats >700 mg Cl^-/L increased the abundance of *Culex restuans*, a mosquito vector for West Nile virus. They showed that the loss of competitors (primarily cladoceran zooplankton) combined with a high salinity tolerance was primarily responsible for the increase in *C. restuans*. Road salt can also decrease the time to emergence for mosquitos (Schuler & Relyea, 2018). As such, road salt contamination may lead to an increase in recruitment of salt-tolerant mosquito species through reduced competition with zooplankton, which may lead to an increased prevalence of mosquito-transmitted diseases, but this needs further study. The possibility that road salt contamination may lead to elevated disease transmission is troublesome particularly given relatively recent issues surrounding diseases such as Zika virus and West Nile virus (Marini, Guzzetta, Rosa, & Merler, 2017; Newman et al., 2017).

Collectively, research from wetlands provides some of the greatest insights into how road salt alters competitive and predator–prey interactions to change community structure. We agree with Petranka and Doyle (2010) that further investigation is needed to determine how road salt in wetlands leads to the export of mosquito vectors. If this unintended consequence is occurring across vast regions where road salt is applied, it is possible road salts are increasing the prevalence of disease transmission to humans due to altered community structure in wetlands. Further, temperature has strong effects on the physiology of freshwater organisms. Since road salts can interact with temperature (Silver et al., 2009), it is also essential future research examine how climate change (e.g. higher spring temperatures, reduced ice cover) might interact with road salt salinisation in wetlands and other freshwater systems.

4 | ECOSYSTEM-LEVEL RESPONSES

Road salt can alter nutrient and carbon cycles, but our understanding of how these cycles are affected are limited to a few studies. Previously, we mentioned that road salt contamination in lakes can alter hydrology and oxygen dynamics and lead to more phosphorus being released from lake sediments. Road salt can also increase the leaching of soil nutrients (Mg^{2+}, K^+, Na^+), potentially inhibiting the recovery of forested catchments affected by acid rain-induced soil acidification (Schweiger, Audorff, & Beierkuhnlein, 2015). Road salt concentrations ranging from 500 to 10,000 mg Cl^-/L in urban watershed networks can also cause the release of Ca^{2+}, K^+, dissolved organic carbon, dissolved inorganic carbon, total dissolved Kjeldahl nitrogen (as NH_4^+ + NH_3 + dissolved organic nitrogen), and can increase the water column concentration of soluble reactive phosphorus released from freshwater sediments or soils (Duan & Kaushal, 2015; Haq, Kaushal, & Duan, 2018). Concentrations of 2,500 mg Cl^-/L in stream ecosystems can reduce denitrification rates, potentially decreasing the number of nitrate sinks in streams (Hale & Groffman, 2006). Thus, in addition to the species- and community-level effects, road salt can disrupt nutrient and energy flow in contaminated stream watersheds through the mobilisation of nitrogen and carbon (Figure 3; Duan & Kaushal, 2015).

Road salt can also alter ecosystem processes in freshwater wetlands. As in streams, road salt can inhibit denitrification (Lancaster et al., 2016). Contamination of sediments from road salts can increase microbial mats at the sediment–water interface, leading to a reduction in pH, but elevated concentrations of exchangeable cations (Ca^{2+}, Mg^{2+}, K^+, Na^+), Fe(II), and Mn(II), which may increase primary production and anaerobic respiration (Kim & Koretsky, 2013). Anaerobic respiration in wetlands is the largest natural contributor of methane in the atmosphere (Matthews & Fung, 1987). Because road salts increase anaerobic respiration (Kim & Koretsky, 2013), contaminated wetlands could emit more methane to the atmosphere,

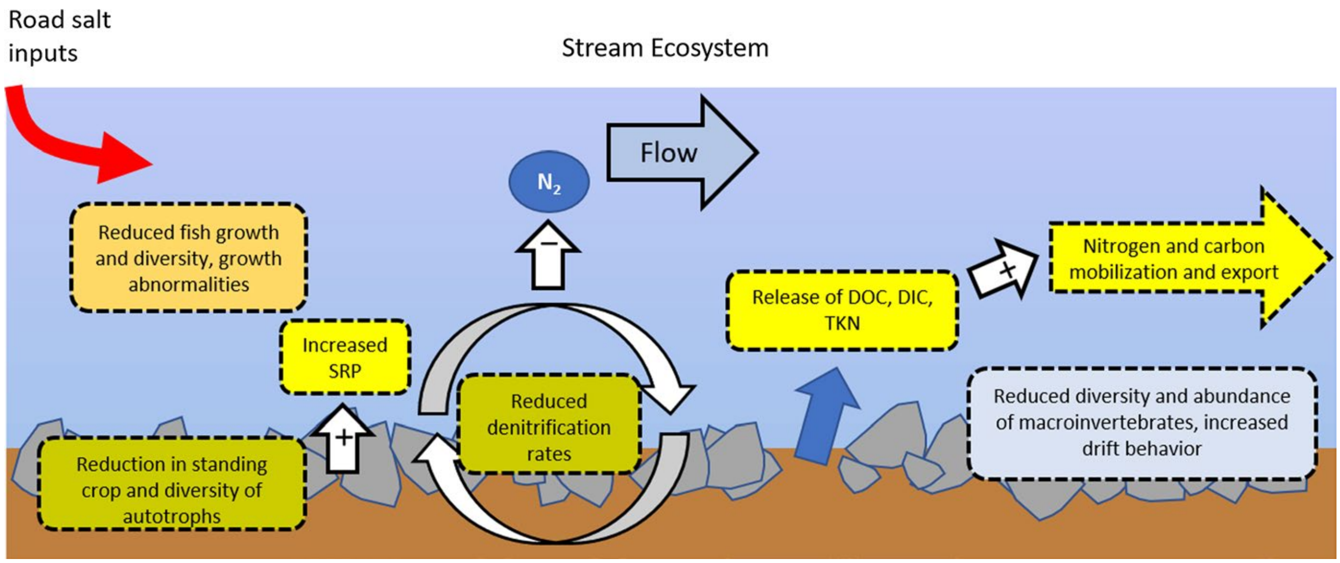

FIGURE 3 Summary of the species-, community-, and ecosystem-level impacts of road salts on stream ecosystems. A reduction in the diversity, abundance, and growth of stream organisms will lead to communities of salt-tolerant organisms. Although reduced rates of denitrification will decrease nitrogen export as N_2, greater export of dissolved organic carbon (DOC), dissolved inorganic carbon (DIC), and total dissolved Kjeldahl nitrogen (TKN; as ammonium [NH_4^+] + ammonia [NH_3] + dissolved organic nitrogen) will occur. SRP: soluble reactive phosphorus [Colour figure can be viewed at wileyonlinelibrary.com]

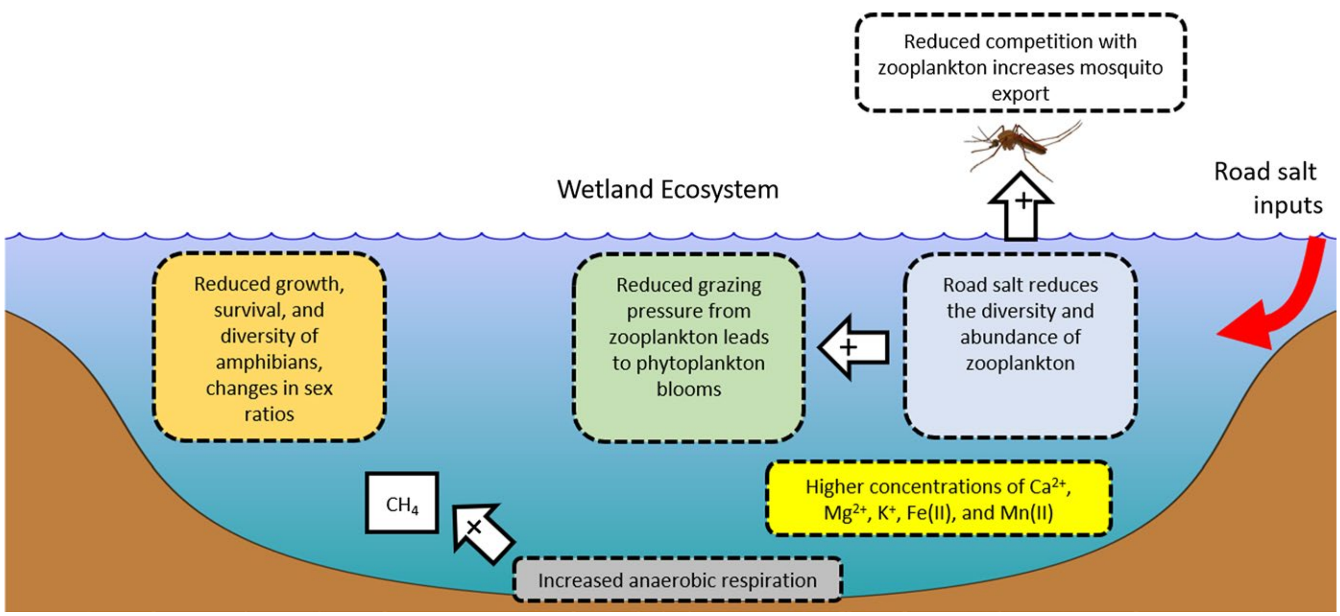

FIGURE 4 Species-, community-, and ecosystem-level impacts of road salts on wetland ecosystems. Reductions in zooplankton abundance and diversity will increase phytoplankton concentrations and through reduced competitive interactions, might export disease vectors like mosquitos. Declines in amphibian diversity will occur from reductions in growth, survival, and altered sex ratios. Road salts lead to more anaerobic respiration and potentially greater exports of greenhouse gasses such as methane (CH_4) [Colour figure can be viewed at wileyonlinelibrary.com]

potentially exacerbating the greenhouse effect on earth's climate. Ultimately, the ecosystem-level impacts of road salts—combined with the species- and community-level impacts (Figure 4)—could potentially lead to regime shifts in contaminated wetlands.

There are many conspicuous gaps in ecosystem-level responses to road salt salinisation. We do not have a robust understanding of how road salts will alter processes such as ecosystem metabolism, oxygen dynamics, nutrient cycling (e.g. phosphorus), carbon cycling, or remineralisation. We have discussed a few biogeochemical responses to road salt, but we do not know how such responses will alter ecological stoichiometry in freshwater systems. Surprisingly, while few studies have examined ecosystem responses to road

FIGURE 5 Summary diagram of the impacts of road salts at three ecological scales

salt in streams and wetlands, studies in lakes are lacking. Linking up ecosystem-level responses with species- and community-level responses will be critical for understanding whole-ecosystem responses to road salt salinisation.

5 | CONCLUSION

Our review indicates that road salt salinisation is altering freshwater ecosystems at multiple levels of biological organisation through reduced growth and abundance of sensitive species, reduced biodiversity, altered community structure, and changes in ecosystem function (Figure 5). Throughout, we have indicated multiple specific areas in need of further research. More broadly, we do not understand how road salt salinisation is altering freshwater ecosystems across much of the world where they are applied, particularly Asia and parts of Europe. We also do not yet understand how other major threats to freshwater ecosystems, such as cultural eutrophication or invasive species will interact with road salts. Both issues affect the same ecosystems in large portions of colder regions throughout the world and a multi-stressor approach to the study of road salt salinisation is needed. Moreover, a topic we did not cover in this review is evolutionary responses to road salt. Relatively recent research shows some freshwater species are able to adapt to road salt salinisation (Brady, 2012, 2017; Hintz et al., 2019). It is important to understand how rising salinity from road salt is shaping species evolution and identify the ecological outcomes (Brady, Monosson, Matson, & Bickham, 2017; Brady & Richardson, 2017; Brady et al., 2017).

The protection of freshwater ecosystems from the unintended contamination by road deicing salts is a difficult task because human safety is undoubtedly a larger priority. However, as we have shown, road salts will likely affect valuable ecosystem services. At present, the task of balancing human safety with ecological impacts may lie in deicing practices. It is critical that policy makers, environmental managers, transportation professionals, scientists, and the public recognise that human safety comes at a cost of changing the structure and function of our freshwater ecosystems. These costs may result in a loss of valuable ecosystem services such as water clarity, drinkable water, recreation venues, and fisheries. Therefore, it is important to find innovative ways of keeping humans safe on the road during the winter months, but at the same time protecting freshwater ecosystems from road salt salinisation.

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