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Ecology of herpetofaunal populations in smelting tailings wetlands

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Abstract. Since the 1920s, Sudbury, Ontario, Canada has emerged into one of the world's largest metal producers. The mining and smelting industries have left a devastating ecological footprint on the Sudbury landscape with metal-contaminated substrates and acidified waters near the smelting facilities and in tailings wetlands. We examined whether the perturbations caused by smelting activities have had a negative effect on ecological aspects of amphibian and reptile populations by investigating differences in richness, abundance, and biomass among three tailings wetlands and a non-tailings reference wetland. Day and night field surveys were performed 2-3 times per week for an entire active season (22 May – 24 September, 2009). We sampled amphibians and reptiles using visual surveys, cover boards, and anuran calls. Although no differences among sites in species richness were detected, the three tailings wetlands demonstrated lower abundance and biomass than the non-tailings wetland. Furthermore, our findings suggest that the tailings wetlands may not be able to sustain the large herpetofaunal community present at the non-tailings wetland. Our study suggests that the effective recovery of tailings wetlands for sustainable amphibian and reptile communities requires additional rehabilitation in order to establish higher trophic levels.

Keywords. Diversity, Abundance, Biomass, Acidification

Introduction

Smelting industries are important sources of pollution and toxins in the natural environment (Telmer et al., 2006). The mining and smelting industries have long been part of Sudbury, Ontario's history (Snucins et al., 2001) and Sudbury has developed into one of the world's largest copper and nickel producers (Winterhalder, 1995). In the 1960s, when mining and smelting were in full production, over 2,000,000 tonnes per annum of sulphur dioxide (SO_2) emissions and 1,500 tonnes of metal particulates per annum were released into the atmosphere and deposited onto Sudbury's regional waters and soils (CEM, 2004). The soils and lakes closest to the smelters (approximately 30 km away) were the most severely damaged by acidification and metal contaminants; however, over 7,000 lakes were impacted within a 17,000 km^2-area surrounding Sudbury (Dixit et al., 1992; Matuszek et al., 1992). During the 1990s, abatement measures were implemented and the smelting industry reduced its SO_2 emissions by approximately 90% (Shuhaimi-Othman et al., 2006; SARA, 2008). Regardless of the dramatic reduction, the mining and smelting industries have left a damaging and lasting ecological footprint on the Sudbury region (Winterhalder, 1995). In addition to damage to these habitats in the "fallout zone" around the mining operations, habitats are also impacted and modified on-site. For example, tailings are the sulphuric acid-based pyrrhotite waste by-products of the milling process of ore. Tailings are layered on the ground and covered with clayey soil and water to neutralize the acid (Golder, 2007); the reclaimed tailings areas are then considered tailings wetlands. Although the levels of contamination in the soils and lakes near the smelting complexes have been previously examined (Snucins et al., 2001; Keller et al., 2003), little attention has been given to the ecology of on-site tailings wetlands.

Amphibians and reptiles are often a diverse but cryptic component of an ecosystem (Vitt et al., 1990; Goldstein et al., 2005; Watling and Ngadino, 2007) and can thus serve as excellent bioindicators of stressed ecosystems (Gibbons et al., 2000; Venturino et al., 2003). Herpetofauna have certain key lifestyle components that make them excellent bioindicators: (1) their aquatic and terrestrial life stages means they are in direct and constant contact with their environment (Vitt et al., 1990; Lips, 1998); (2) because they are ectotherms (MacCulloch, 2002), they rely on environmental conditions to maintain their metabolism and other life processes (Feder and Burggren 1992; Pough et al., 2003); (3) amphibians exchange gases through their moist, semi-permeable skin, which plays

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a role in chemical uptake (Brattstrom, 1979), while, reptiles are better protected against the environment as their integument is covered with scales (Gibbons et al., 2000), but their eggs, like those of amphibians, are still susceptible to metal contaminants (Marco et al., 2004). Both orders are susceptible to acidification (Vitt et al., 1990; Gibbons et al., 2000) and metal contaminants (Christy and Dickman, 2002), and both groups are known to be declining in numbers (Gibbons et al., 2000; Stuart et al., 2004). Thus, studying the population ecology of amphibians and reptiles in tailings wetlands should provide insight into the health and mitigation potential of such wetlands.

Environmental and anthropogenic factors that influence amphibian and reptile population declines can be examined through population ecology studies. For example, diversity measures such as species richness, defined as the number of species present in a given area, and abundance, defined as the number of individuals within a specific community, are indicative of community and population structures within a specific ecosystem (McArthur and Wilson, 1974). The objectives of our study were to compare species richness, abundance, and biomass of amphibians and reptiles between tailings wetlands and a non-tailings reference wetland, and to use our findings to make recommendations for recovery of tailings wetlands at smelting sites. We hypothesized that because tailings wetlands are created from neutralized sulphuric acid mining by-products, they constitute unsuitable habitat for amphibians and reptiles. We thus predicted lower species richness, abundance, and biomass in the tailings wetlands as compared to the non-tailings site.

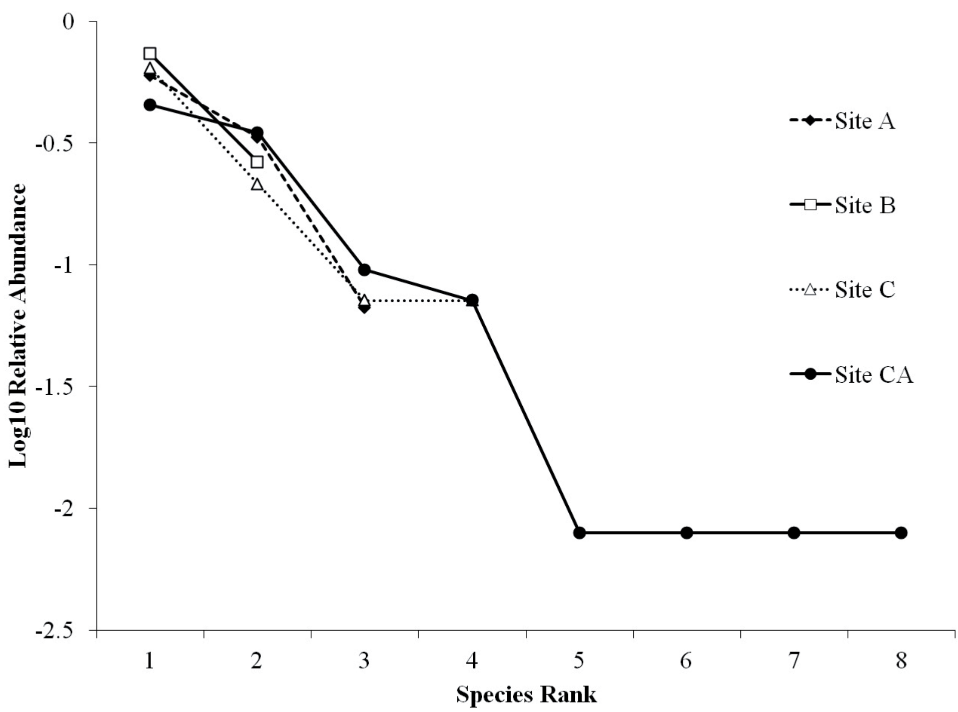

Figure 1. A rank abundance plot visually explains four aspects of species richness. (1) Each dot on a line represents a species within the specific sample site; (2) the lines demonstrate the log relative abundance levels of species at each of the four sample sites; (3) each species is placed in descending order in relation to their abundance ranking in that particular site; and (4) as the slope of each line becomes more horizontal, it implies that the evenness level becomes stronger within each specific site. The relative abundance axis depicts the abundance levels for each species. The species rank is described as a dominance ranking for each species on an ordinal scale, which means that the lower rank number (e.g. 1) indicates a higher ranked or more abundant species. Sites A, B and C are the tailings wetlands studied at Xstrata Nickel, Falconbridge, Ontario, and Site CA is the non-tailings wetland studied at Laurentian Conservation Area, Sudbury, Ontario.

Table 1. The surface areas (m^2) of each sample site were calculated using ArcGIS and Ontario Base Maps (www.geographynetwork.ca/website/obm/viewer.htm). The vegetation and water characteristics of each sample site were interpreted by visual observations. The vegetation characteristics include the dominant type of flora present at the water's edge and approximate distance of forest from water's edge. Water characteristics were denoted by the maximum depth, wetland type (identification based on Lyons (2003)), and residues observed on the water surface. Sites A, B and C are the three tailings wetlands studied at Xstrata Nickel, Falconbridge, Ontario, and site CA is the non-tailings reference wetland studied at Laurentian Conservation Area, Sudbury, Ontario.

		Vegetation characteristics		Water characteristics	
Sample site	**Surface area (m^2)**	**Water's edge**	**Distance to forested area**	**Depth (m)**	**Wetland type**
A	5469.2	Short Emergent Grasses	Near wetland	1.4	Pond
B	12665.0	Cattails and Tall Emergent Grasses	Near wetland	0.8	Pond
C	5794.3	Tall Emergent Grasses	Far from wetland	1	Marsh
CA	5001.2	Tall Emerging Grasses	Far from wetland	1.2	Pond

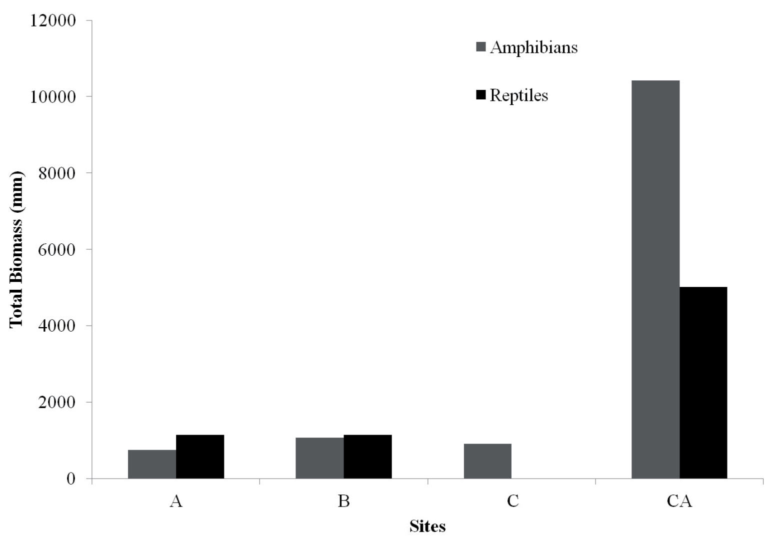

Figure 2. Total biomass of amphibians and reptiles per sample site measured with respect to average body length (mm). For every sample site, the biomass was calculated by multiplying the relative abundance of each species with the average body length for each species present. Sites A, B and C are the tailings wetlands studied at Xstrata Nickel, Falconbridge, Ontario, and Site CA is the non-tailings wetland studied at the Laurentian Conservation Area, Sudbury, Ontario. $\chi 2$ analyses demonstrated that Site CA had significantly higher values in all four biomass analyses compared to the three impacted wetlands (Amphibian Biomass P <0.001; Reptile Biomass P <0.001; Total Biomass = P < 0.001).

Methods

Sampling Sites

Amphibian and reptile surveys were conducted from 22 May 2009 – 24 September 2009 at four sites. Three tailings wetlands were located on the property of Xstrata Nickel in Falconbridge, Ontario, Canada; Sites A (N 51°59.732; W 51°54.156), B (N 51°60.923; W 51°62.132), and C (N 51°60.292; W 51°72.484). The fourth site was a non-tailings reference wetland located at the Laurentian Conservation Area, Sudbury, Ontario, Canada; Site CA (N 46°27.630'; W 80°56.449'). All wetlands were similar in size, water depth, and vegetative composition (Table 1). Geographic Information System (GIS) software (ArcGIS 9.0, Redlands, CA) was used to make cartographic maps of each of the four sites. The surface area of each site (m^2) was determined though polygon mapping and analysis. Because each wetland differed in surface area, a correction factor ([[variable/surface area]/search effort] x 1000 m^2) was used to standardize species richness and abundance variables by the size of the wetlands to allow us to make comparisons among sites.

Field Surveys

Surveys were performed 2-3 times per week by two people. Each site was surveyed between 0.70 – 1.29 h, depending on the size of the wetland. Morning, afternoon, and evening survey times were alternated throughout the study period. Each survey included multiple components. First, a visual scan with binoculars was performed at the water's edge and amphibians and reptiles were identified to species. Second, surveys along wetland edges were conducted on foot. In Sites A, C and CA, the entire wetland area was surveyed. In Site B, only half of the pond area was surveyed due to the large size of the wetland. In addition, certain areas of Site B were deemed unsuitable habitat for the studied species and were not surveyed as these areas were abundantly populated with cattails (*Typha* spp.; Lacki et al., 1992). Third, cover boards were sampled once per survey, either before or after the on foot surveying. On the south side and in a southern direction, a linear array of seven plywood cover boards (50 cm x 80 cm; deployed in May 2009) were placed 10 m apart, beginning at the wetland edge and running perpendicularly into the forested area surrounding each wetland; GPS locations were recorded for each cover board. Fourth, anuran calls were recorded during the visual surveys. Lastly, habitat characteristics were recorded at each site (Table 1), including dominant vegetation type at the wetland edge, distance to surrounding forest, maximum water depth, and type of wetland categorized using a wetland identification handbook (Lyon, 1993). Vegetation around the periphery of the wetland was classified by the main plant category (e.g. tall emergent plants). Distance to surrounding forest was defined as the approximate distance to the forest from the water's edge in two categories: Near (0-5 m) and Far (10 – 15 m). Maximum water depth was measured (± 1 cm) at one location in the deepest part of the wetland using a metric tape measure.

Daytime Surveys

One researcher surveyed the water's edge while the other surveyed the terrestrial area surrounding the wetland (grassy/forested area up to 8 m from the wetland's edge). The terrestrial survey was implemented to search for amphibians and reptiles that reside in the terrestrial portion of the wetland (e.g. salamanders, snakes and some frog species). The sampling procedure occurred in five steps. First, individuals were either caught by hand or by dip net. Second, captured specimens were identified to species using the Royal Ontario Museum Field Guide to Amphibians and Reptiles of Ontario (MacCulloch, 2002) or the Toronto Zoo Amphibians, Turtles and Snakes of Ontario Identifier (Toronto Zoo, 2009). Third, each specimen was photographed with a digital camera so that photo identification could be used as a non-invasive mark and recapture technique (Plăiaşu et al., 2005; Speed et al., 2007). In addition, turtles were given a permanent identification code using shell notching (Cagle, 1939; Gamble and Simmons, 2004). Fourth, body length was measured using vernier dial calipers (model F13416-0001, Scienceware, Garner, NC). Anurans were measured from the tip of the nose to the end of the ileum bone (snout-to-vent, SVL); snakes and salamanders were measured from the tip of the nose to the end of tail (total length, TL); and turtles were measured from the nuchal scute to the caudal end of the carapace (midline carapace length, CL). Fifth, age categories were assigned using different methods for each group. To demarcate age classes of anurans, we used SVL (Wood et al., 1998); if SVL was within the adult range for the specific anuran species, they were classified as an adult (Harding, 1997; Wright and Wright, 1967). For turtles, if carapace length was within the adult range for the specific species, they were classified as an adult (MacDonald and Metcalfe, 1991). For snakes, body length was used to identify age category; if the total body length was within the adult range for the specific species, they were classified as adults (Toronto Zoo, 2009).

Data Handling and Statistical Analyses

Data handling consisted of a three step process: 1) sorting all the data files for the marked amphibians and reptiles; 2) filtering out the recaptured individuals by shell notching for turtles or by examining each specimen against the collection of digital images; and 3) assigning individuals to adult or juvenile age classes. Only individuals that were considered adults were used in the statistical analyses.

Species diversity is explained through two components: species richness and evenness. Species richness included three separate analyses: total species richness, amphibian species richness, and reptile species richness. Simpson's evenness index was implemented to measure evenness (Odum, 1975; Magurran, 2004). Biomass (defined as biolength because we did not measure body mass) was calculated using the average body lengths (ABL) of adults for each species, which were acquired from Wright and Wright (1967), total abundance of each site (TA), and the relative abundance (RA; calculated by dividing the abundance of each species by TA). The formula for the biomass calculation was [(RA*n)*ABL] where "n" is the total number of individuals of all species. We examined biomass in three ways: total biomass, amphibian biomass, and reptile biomass at each of the four sites. Chi-square tests were performed to determine whether there were differences among sites in total species richness, amphibian species richness, reptile species richness, abundance, total biomass, amphibian biomass, and reptile biomass. However, for the abundance and biomass analyses, certain anuran species (i.e. *Pseu-*

Table 2. List of all the amphibian and reptile species (common and scientific names) found at the four sample sites during the surveying period of 22 May to 24 September, 2009. Sites A, B and C are the three tailings wetlands at Xstrata Nickel, Falconbridge, Ontario, and Site CA is the non-tailings reference wetland at Laurentian Conservation Area, Sudbury, Ontario.

		Species Presence			
Common Name	Scientific Name	Site A	Site B	Site C	Site CA
Amphibians					
Green Frog	*Lithobates clamitans*	x	x	x	x
Bullfrog	*Lithobates catesbeianus*				x
Leopard Frog	*Lithobates pipiens*	x	x	x	x
Mink Frog	*Lithobates septentrionalis*	x	x	x	x
Wood Frog	*Lithobates sylvaticus*	x	x	x	x
Spring Peeper	*Pseudacris crucifer*	x	x	x	x
Grey Tree Frog	*Hyla versicolor*	x	x	x	x
American Toad	*Anaxyrus americanus*				x
Blue-Spotted Salamander	*Ambystoma laterale*				x
Reptiles					
Northern Red-Bellied Snake	*Storeria occipitomaculata occipitomaculata*	x	x	x	x
Garter Snake	*Thamnophis sirtalis sirtalis*			x	x
Smooth Green Snake	*Opheodrys vernalis*				x
Northern Ribbon Snake	*Thamnophis sauritus septentrionalis*	x			
Water Snake	*Nerodia sipedon*				x
Painted Turtle	*Chrysemys picta*				x
Snapping Turtle	*Chelydra serpentina*				x

dacris crucifer, *Lithobates sylvaticus* and *Hyla versicolor*) were not included as they are not easily captured due to their camouflage and avoidance abilities. All statistical analyses were performed using Statistica 8.0.360 (StatSoft, 2007, Tulsa, OK).

Results

Species Richness and Evenness

We identified a total 15 species from the capture and call data at the four sample sites (Table 2). A core assemblage of species was present in all sites: Green Frog (*Lithobates clamitans*), Leopard Frog (*Lithobates pipiens*), Mink Frog (*Lithobates septentrionalis*), Wood Frog (*Lithobates sylvaticus*), Grey Tree Frog (*Hyla versicolor*), Spring Peeper (*Pseudacris crucifer*), and Red-Bellied Snake (*Storeria occipitomaculata occipitomaculata*). Other species were present in only specific sites. Site A was home to the Northern Ribbon Snake (*Thamnophis sauritus septentrionalis*) and Site C was home to the Eastern Garter Snake (*Thamnophis sirtalis sirtalis*) in addition to the core assemblage. At Site CA, along with the core assemblage, we observed Bullfrog (*Lithobates catesbeianus*), American Toad (*Anaxyrus americanus*), Blue-Spotted Salamander (*Ambystoma laterale*), Eastern Garter Snake, Smooth Green Snake (*Opheodrys vernalis*), Painted Turtle, Snapping Turtle (*Chelydra serpentina*), and Northern Water Snake (*Nerodia sipedon*). Despite these different community compositions, there were no significant statistical differences in species richness or evenness among sites (Table 3). Nonetheless, the tailings sites showed a trend for lower species richness per m^2 (Site A = 1.85; B = 0.80; and C = 1.88) compared to the reference site (CA = 2.32) and the Simpson's Index indicated a lower, but not significant, evenness at the impacted sites (Site A=0.285; B=0.212; C=0.292) compared to the reference site (CA=0.372). The tailings sites exhibited relatively lower species richness, but the species that populated the sites were present in relatively high abundance (Figure 1).

Abundance

In general, the tailings sites supported one or two dominant species and low abundances of other species, as indicated by the steeper slope in Figure 1 for all the tailings sites (A, B and C) as compared to the non-tailings site (CA). Within the four sites, we captured a total of 174 individual adults. At Site A, we captured 8.6% of the total of individuals, which consisted of three species (i.e. Green Frog, Mink Frog, and Red-Bellied Snake). At Site B, we captured 10.9% of the total of individuals including Green Frogs and Red-Bellied Snakes. At Site C, 8.0% of the total was captured consisting of four species: Green Frogs, Mink Frogs, Leopard Frogs, and Wood Frogs. Site CA had the largest percentage (72.4%) of individuals caught consisting of eight species: Green Frog, Bullfrog, Leopard Frog, Blue-Spotted Salamander, Red-Bellied Snake, Northern Water Snake, Painted Turtle, and Snapping Turtle. Amphibian and reptile abundance within the reference wetland was significantly higher than in the three tailings wetlands (Table 3).

Biomass

In every measure of biomass, the reference site had a significantly higher biomass than the three tailings sites (Table 3). Total biomass was significantly different among sample sites (Figure 2). Similarly, both amphibian and reptilian biomass were significantly different among sites (Table 3).

Discussion

Species Richness and Evenness

Our prediction that species richness would be lower in the tailings wetlands was not supported by our data as no significant differences were found among sites for total species richness, amphibian species richness, or reptile species richness. However, the tailings sites did not have the same species assemblage as the reference site: Snapping Turtles and Painted Turtles are higher trophic level species that were not found at the tailings sites in our study. Turtles need high levels of resources in order to sustain optimal energy levels to invest in reproduction (Congdon and Tinkle, 1982) and our findings suggest that the impacted sites did not provide adequate resources for turtles. Furthermore, we observed two different top anuran predators at separate sites. Bullfrogs were present in the reference site but absent at all three tailings sites and Mink Frogs were abundant at tailings Site C but rare at the reference site. Bullfrogs are known to out-compete Mink Frogs and many other anuran species (Hayes and Jennings, 1986; Werner and McPeek, 1994), possibly explaining why we did not observe many Mink Frogs at the reference site. At the tailings sites, there were no Bullfrogs, probably because their egg clutches are highly sensitive to acidic and metal contaminated wetlands (Freda et al., 1991). Because both Mink Frogs and Green Frogs are more acid tolerant than Bullfrogs (Pierce, 1985), either species may become the dominant anuran species at impacted sites where Bullfrogs are absent (Stewart and Sandison, 1972; Hecnar and M'Closkey, 1997). These differences in species compositions among sites suggest that the tailings sites may not be able to sustain higher aquatic trophic levels and top predators.

Our tailings wetlands study sites may be receiving acidic inputs from the atmosphere and from nearby tailings wetlands via underground leaching and inadequate capping or lining of tailings. Acidification is caused by the deposition of atmospheric pollutants resulting from activities such as smelting, mining, and fabrication (Keller et al., 2007), and can negatively impact amphibian and reptile species richness. Wyman and Jancola (1992) observed that acidic breeding sites had lower species richness as compared to a non-acidic breeding site. Other studies have demonstrated the effects of acidification on amphibian embryonic and larval stage mortality rates (Carey and Bryant, 1995). Freda et al. (1991) observed that Bullfrog and Leopard Frog embryonic and larval stages were highly sensitive to high acidic levels. Although we did not quantify acidic inputs from underground leaching, this could be a reason for the absence of Bullfrogs within the tailings sites as Bullfrogs are sensitive to acidic environments (Freda et al., 1991).

Within an amphibian and reptile assemblage, extreme evenness levels among species can help explain abundance and biomass levels. Rank abundance plots are used to illustrate species dominance within a community hierarchy. In this context, the degree of dominance is determined by the species' abundance level within a specific community. Species with a higher dominance will have higher abundance levels and ranking designation when plotted. Although no significance was detected, the tailings sites presented fewer species and the abundance levels of each species decreased in proportion to their assigned rank, indicating lower evenness levels. By contrast, the reference site demonstrated a greater number of species, where the dominant ranking species exhibited higher abundance levels than the lower ranking species. This reference site accommodated a higher level of species richness and evenness than the tailings sites (Figure 1) because the reference wetland is ecologically more sustainable allowing for more diverse and abundant community. These results suggest that higher levels of evenness, where there are varying degrees of dominance among species, favour higher abundance, and biomass within a wetland. Similarly, studies have shown that acidic peatlands have lower levels of amphibian diversity, species evenness and abundance (Karns, 1992). Other studies have suggested that non-tailings wetlands that have sufficient resources sustain increased dominance as interspecific competition is high, and species evenness will likely decrease (Nijs and Roy, 2000). These short-term studies indicate that hierarchal dominance levels within a community assemblage are indicative of a highly sustainable wetland habitat.

Abundance and Biomass

As predicted, abundance levels were significantly different among sites. The reference site had significantly more individuals/m^2 than the three tailings sites. Similarly, amphibian and reptile biomasses were significantly higher at the reference site than in the three tailings sites, thus supporting our hypothesis that tailings wetlands are unsuitable habitat for amphibians and reptiles. The acidity and toxins within these tailings wetlands may be an explanation for these negative impacts on amphibian populations (Carey and Bryant, 1995; Alford and Richards, 1999). Amphibians breed in temporary ponds, which are fairly small and shallow bodies of water that receive their water by rainfall, runoff or snowmelt (Wiggins et al., 1980). These ponds can be affected by acidic deposition as well as runoff from contaminated nearby areas (Freda et al., 1991). Sudbury's lakes have been severely impacted by acidification and metal pollutants and most if not all water bodies are still suffering from some low to moderate effects of acidification (Snucins et al., 2001). As tailings wetlands are built upon acidified tailings, there is a reasonable chance of having higher concentrations of acidity present within the water (Schwab et al., 2007). In response to stressed environments, amphibians make trade-offs in growth rates, size, and age at maturity in order to optimize their fitness levels (Lefcort et al., 1998; Merilä et al., 2004). Acidic and poor conditions tend to reduce growth rates and development, especially in seasonal and temporary habitats such as breeding ponds used by amphibians (Räsänen, 2002). In fact, studies have indicated that increased metal contamination (Fahrig et al., 1995; Christy and Dickman, 2002; Sanzo and Hecnar, 2006) and acidification levels (Corn and Vertucci, 1992; Carey and Bryant, 1995; Räsänen, 2002) negatively affect growth rates and disrupt normal development of larval, juvenile, and adult anurans. Further, female amphibians originating from acidic habitats tend to invest in larger eggs rather than larger clutches (Räsänen, 2002), which could result in lower amphibian abundance levels in contaminated areas. Finally, the acid levels of contaminated breeding ponds and streams are detrimental to the survivability of salamanders (Kucken et al., 1994) which could explain why we did not observe salamanders at the tailings sites. Another explanation for the observed difference in biomass among sites could be the fact that amphibian growth depends on environmental factors such as temperature, food availability and water (Pough et al., 1992). Diet quality is positively correlated with growth (Jorgensen, 1992) suggesting that the tailings sites do not have sufficient resources for anurans to reach the biomass levels observed in the non-tailings reference site.

It is possible that our observed significant differences in population ecology between tailings and non-tailings wetlands could be explained by habitat differences among sites rather than being the consequence of smelting related activities. Amphibians and reptiles rely on more than the aquatic habitat for their daily activities within a wetland ecosystem. Snakes, such as Eastern Garter Snakes, Red-Bellied Snakes, and Ribbon Snakes, which were all present in the tailings wetlands, inhabit and require a dense grassy wetland terrain to effectively hunt, travel, and avoid predation and desiccation from hot conditions (Carpenter, 1952); while, anurans

Table 3. Calculated values and results of chi-square analyses for ecological variables compared among four sample sites in the study if the impacts of mining tailings on herpetofauna. Sites A, B and C are three tailings wetlands at Xstrata Nickel, Falconbridge, Ontario, and Site CA is a non-tailings reference wetland studied at the Laurentian Conservation Area, Sudbury, Ontario. See Methods for variable definitions.

Variables	Site A	Site B	Site C	Site CA	χ^2	P value
Species richness	1.86	0.80	1.88	2.32	0.73	0.86
Amphibian species richness	1.39	0.68	1.26	1.39	0.29	0.96
Reptile species richness	0.46	0.11	0.42	0.93	0.71	0.87
Evenness	0.29	0.21	0.29	0.37	0.45	0.99
Abundance	3.95	1.82	1.87	19.53	32.26	<0.001
Total biomass (mm)	1887.5	2206.50	905.00	15431.50	28003.27	<0.001
Amphibian biomass (mm)	745.00	1064.00	905.00	10422.00	20702.08	<0.001
Reptile biomass (mm)	1142.5	1142.50	0.00	5009.50	7898.16	<0.001

require either specific aquatic or terrestrial conditions depending on the species. For example, Green and Mink Frogs typically reside in aquatic settings, where the water surface area, depth, and vegetation in and around the water's edge plays a crucial role in their habitat requirements. In contrast, Spring Peepers, Wood, Grey Tree, and Northern Leopard Frogs spend more time in the terrestrial vegetation surrounding the wetland, where the vegetation and forested area surrounding the wetland are important habitat requirements (Hecnar, 2004). In our study, all four wetlands were chosen because they contained similar habitat characteristics and would be suitable wetlands for amphibian and reptile populations. Wetland surface area differed among sites, and this could affect herpetofaunal diversity, abundance, and biomass. For instance, Site B is twice as large as the other sites. With a larger wetland surface area, diversity and abundance should increase (Rusell et al., 2002). However, only part of Site B was sampled, and we standardized our data to account for differences in surface area and search effort prior to analyses. Furthermore, although only Site B was significantly larger in size, all three tailings sites had greater surface areas than the reference site, but still yielded lower abundance and biomass estimates compared to the reference site, lending support to our conclusion that tailings wetlands cannot support the diverse herpetofaunal community of the reference site. Additionally, we observed a notable visual difference in water quality in the tailings sites compared to the reference site. All of the former presented petrol residues and Site B had large sections with a thick layer of iron residue. There were no distinguishable wetland characteristics in terms of size, depth, and surrounding habitat to support our observed differences in abundance and biomass; however, although water chemistry was not performed in this study, perhaps it could potentially help explain our findings.

Herpetofaunal populations demonstrate large fluctuations in annual and seasonal population size and abiotic variability (Pechmann et al. 1991; Gibbons et al., 1997). Since this study is considered short-termed (one year) and spatially limited (four sample sites), our results can only accurately suggest that presently, the tailings wetlands we sampled are not suitable habitat for herpetofauna. Additionally, capture probabilities can be highly variable as a result of local causes that consistently affect herpetofaunal population abundance. These population fluctuations occur not only annually but throughout the breeding season (Gibbons et al., 2000; Green, 2003). Anuran and reptile ecology is sensitive to a large variety of factors that can cause large variability within their annual community structure such as pond desiccation, accessibility of water bodies and terrestrial habitat, and heterogeneity of surrounding habitat (Schmidt, 2004). Furthermore, ecologists sometimes fail to take into account recapture probabilities when estimating amphibian and reptile demographics (Schmidt, 2003). In our study, a mark-capture-recapture method was used to reduce the potential for overestimating detection probabilities in these disturbed populations. Additionally, the capture procedure was standardized allowing for recaptured individuals to only be assessed once during the statistical analyses. As this project was spatially restricted, the value of this abundance information may be limited; however, it is still fundamentally critical in understanding how metal-stressed sites affect herpetofaunal assemblages, and serves as a starting point for future work across greater spatial and temporal scales.

Conclusion

The results from our short-term study suggest that tailings wetlands do not adequately support ecological aspects of amphibian and reptile populations in terms of abundance levels and community structure. Although there were no significant differences found among sites in species richness, abundance and biomass levels were significantly lower in the tailings sites compared to the reference site. These findings may indicate that amphibians and reptiles are still being affected by trace leaching of sulphuric acid and metal contaminants from the buried tailings. This is corroborated by the finding that although similar species inhabit the tailings and reference wetlands, they are present in lower numbers in the tailings sites. The tailings wetlands in our short-term study appear to be in need of rehabilitation in order to establish higher trophic levels and sustainable amphibian and reptile communities. Further, our findings can be used to make recommendations for the rehabilitation of tailings wetlands. For example, a resource availability study in comparison to the reference site would specify what types of ecological resources are insufficient for proper population sustainability. Lastly, an extensive water and groundwater chemistry analysis could indentify if the metal and acid leaching is occurring within the tailings and impacting the resident communities.

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