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Bycatch mortality can cause extirpation in four freshwater turtle species

JONATHAN D. MIDWOOD[a,*], NICHOLAS A. CAIRNS[a,b], LAUREN J. STOOT[a,b], STEVEN J. COOKE[a,c] and GABRIEL BLOUIN-DEMERS[b]
[a]*Fish Ecology and Conservation Physiology Laboratory, Department of Biology, Carleton University, Canada*
[b]*Herpetology Laboratory, Department of Biology, University of Ottawa, Canada*
[c]*Institute of Environmental Science, Carleton University, Canada*

ABSTRACT

1. Bycatch of non-target species in commercial fishing nets can have adverse impacts on their populations. Freshwater turtle populations are particularly susceptible to increases in adult mortality, and freshwater turtles are among the most threatened vertebrates.
2. As a case study, the population-level impacts of bycatch mortality on freshwater turtles were evaluated in Lake Opinicon, Ontario, Canada, a lake that supports a small-scale commercial fishery. Using population viability analyses, the impacts of bycatch on common snapping turtles (*Chelydra serpentina*), eastern musk turtles (*Sternotherus odoratus*), northern map turtles (*Graptemys geographica*), and painted turtles (*Chrysemys picta*) were evaluated.
3. In all four species, even low levels of additional annual female mortality as a result of bycatch were sufficient either to reduce population size or to cause extirpation of the local population within 500 years. Bycatch reduction programmes, such as seasonal closures and implementation of bycatch reduction devices, can help alleviate the risk of extirpation. Changes to fishing season length could help reduce the number of snapping turtles and musk turtles captured. Installation of simple bycatch reduction devices can exclude between 95% and 100% of snapping turtles and between 0% and 97% of the other three species, depending on the width of the exclusion device. If combined, these two bycatch reduction methods would help prevent adult female mortality and help maintain turtle populations in Lake Opinicon.
4. Although these findings are specific to the study area, the same principles apply to other areas where similar simple bycatch reduction strategies can be employed to prevent the extirpation of other freshwater turtle species. Considering the consequences of bycatch and of bycatch reduction programmes on populations provides managers with important information to support development of risk-averse conservation strategies.

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*Correspondence to: Fish Ecology and Conservation Physiology Laboratory, Department of Biology, Carleton University, 1125 Colonel By Dr., Ottawa, ON, Canada, K1S 5B6. Email: midwoodj@gmail.com

INTRODUCTION

Bycatch associated with fisheries is a threat to biodiversity in both marine and freshwater systems (Hall *et al.*, 2000; Raby *et al.,* 2011). The threat posed by bycatch is particularly acute in species that breathe air as soak times of fishing gear typically extend beyond the submergence capacity of most air-breathing species (Melvin *et al.*, 2001; Bury, 2011). Mortality of non-target species captured as bycatch can result in demographic shifts leading to population declines and community changes (Hall *et al.,* 2000; Dorcas *et al.*, 2007). Incorporating capture of non-target species into the management of fisheries is difficult as, by definition, bycatch is unintended. Management challenges are magnified when bycatch is not reported. Bycatch is not reported in many small-scale inland fisheries, making this class of fishery an under-appreciated threat to biodiversity compared with its marine counterpart where bycatch is commonly reported (Raby *et al.,* 2011). Small-scale inland fisheries can be diffuse and isolated making them difficult to regulate, but they are often socio-economically or culturally important to local communities, thereby adding pressure to maintain fish harvest rates (Allan *et al.,* 2005). One method of ensuring the economic sustainability of inland fisheries while preventing capture of non-target species is a bycatch reduction programme (BRP; Hall *et al.,* 2000).

Bycatch reduction programmes are common in many large-scale marine fisheries (Kennelly and Broadhurst, 2002), but seldom used in inland fisheries (Raby *et al.,* 2011). A BRP can be composed of two broad categories of approaches: those that manage effort and those that reduce bycatch per unit effort (Hall *et al.,* 2000). Common examples of these approaches are seasonal or area closures (bycatch reduction strategies; BRSs) and bycatch reduction devices (BRDs), respectively (Melvin *et al.,* 2001). Both methods use differences between target and bycatch species to prevent unintended capture (Hall *et al.,* 2000). Closures rely on regulation or voluntary adoption to reduce the rates of non-target species interacting with fishing gear by spatial, temporal or seasonal avoidance. BRDs are typically modifications that improve the selectivity of traditional fishing gear (Broadhurst, 2000). The use of these two types of BRPs in small-scale fisheries presents logistical (e.g. implementation and enforcement) and economic (e.g. cost of installation) challenges. It would therefore be helpful to determine the risk posed by fishery interactions to populations of bycatch species while also assessing the effectiveness of proposed BRPs at mitigating this risk (Crowder and Murawski, 1998).

Freshwater turtles are cosmopolitan in distribution, composing a substantial part of the freshwater biomass in tropical and temperate regions and filling a variety of trophic niches (Congdon *et al.*, 1986). A reproductive strategy relying on extreme iteroparity renders turtle populations particularly sensitive to the loss of reproductive adults. Furthermore, naturally high mortality at early life stages, delayed sexual maturity, and a lack of density-dependent reproductive responses make turtles ill-equipped to compensate for even low rates of additional adult mortality (Brooks *et al.,* 1991; Congdon *et al.*, 1994). Consequently, about 50% of freshwater turtle species are threatened with extinction, making them one of the groups of vertebrates most at risk (Turtle Taxonomy Working Group, 2012; www.registrelep-sararegistry.gc.ca). Therefore, understanding and managing the potential human impacts on turtle populations is critical. Many threats, including habitat loss, the pet trade, and consumption are evident and well documented. Yet, while threats to marine turtles from bycatch are highly publicized, impacts on freshwater turtles may be less widely touted due to the absence of 'charismatic' species (Campbell and Cornwell, 2008).

Fishery-induced mortality of incidentally captured freshwater turtles has been noted in a number of studies examining fisheries (Sullivan and Gale, 1999; Michaletz and Sullivan, 2002) and turtle/fishery interactions (Bishop, 1983; Barko *et al.*, 2004; Lowry *et al.,* 2004; Fratto *et al.,* 2008a, b). The rates of mortality vary by species (Barko *et al.,* 2004), soak time (Bishop, 1983), and water temperature (Fratto *et al.,* 2008a). A useful tool for assessing the impact of additional mortality on a population is population viability analysis (PVA; Boyce, 1992). PVA uses demographic and reproductive data to model the probability of long-term population persistence in the face of stochastic and/or deterministic trends. It is a useful tool for wildlife and stock management, especially for species where intergenerational trends are hard to observe (Beissinger, 2002; Morris and Doak, 2002). Given their delayed maturity and longevity, population trends for turtles are challenging to document, but PVA has been used to model impacts of additional mortality arising from boat strikes (Bulté *et al.*, 2009). As with any modelling approach, PVAs are limited by the availability of appropriate data and by the unknown responses of a population in novel future scenarios (Coulson *et al.,* 2001). However, PVAs can be useful in conservation science and resource management (Brook *et al.,* 2000; McCarthy *et al.*, 2003) and are widely used in risk assessment of populations of conservation concern (Taylor, 2002).

Here, PVAs are used for four turtle species (snapping turtles (*Chelydra serpentina*), musk turtles (*Sternotherus odoratus*), map turtles (*Graptemys geographica*), and painted turtles (*Chrysemys picta*)) in Lake Opinicon, Ontario, Canada, as a case study to test whether fishing pressure (and the resulting bycatch of adult turtles) can cause population declines in these four species, and whether the addition of simple BRDs to standard gear or management of capture through reduced season length could potentially mitigate the risk posed by bycatch to these four species by reducing capture. This study will help managers make more informed decisions in instances where biodiversity and socioeconomics both warrant consideration.

METHODS

Study site

In eastern Ontario, Canada there is a small-scale fishery that uses passive impoundment style nets (e.g. fyke nets) to collect a variety of panfish (Burns, 2007; Larocque *et al.*, 2012a). This study was conducted on Lake Opinicon, located 100 km south west of Ottawa, Ontario, Canada. Lake Opinicon is a mesotrophic, shallow lake with a mean depth of 2.8 m and a surface area of ~780 ha (Agbeti *et al.,* 1997). This lake supports a small-scale freshwater commercial fishery, characteristic of other lakes in the region, with a single fisher licensed to deploy 80 nets simultaneously. Although turtle bycatch occurs in this type of small-scale commercial fishery (M.A. Carrière pers. comm. Oct. 18, 2013), fishers are not required to report the number of turtles captured or killed (Larocque *et al.*, 2012b; OMNR, 2013). Therefore, catch rates from fishing nets deployed as part of a continuing study of turtle bycatch in Lake Opinicon were used to estimate bycatch in commercial fishing nets (see below).

Population viability analysis

Population viability was modelled in the software Vortex 9.99 (Lacy, 1993, 2000) using species-specific parameters collected from the literature (Supporting information, Table S1). Population viability analysis has previously been used to evaluate the population consequences of bycatch issues in marine mammals (Harwood, 2000; Majluf *et al.*, 2002; Goldsworthy and Page, 2007) and in seabirds (Majluf *et al.*, 2002). In the PVA models, rates of mortality, reproductive age of males and females, and maximum reproductive age were set as fixed variables. However, stochasticity was built into the models through variation in annual clutch sizes and variation in annual number of clutches (see Lacy, 1993, 2000 for a detailed discussion of model assumptions and application in Vortex). Given the lack of a density-dependent reproductive response in turtles (Brooks *et al.*, 1991), this parameter was not incorporated into the PVA models. Similarly, to isolate the effect of bycatch mortality and simplify the models, inbreeding was not included.

Lake-specific life-history parameters of each species could not be calculated owing to the unavailability of long-term demographic data; therefore, an effort was made to use literature on northern populations that should have life histories that are close approximations of the life histories of the study populations (Galbraith, 1986; Iverson, 1991, 1992). The viability of the populations and influence of adult mortality were modelled over 500 years using 1000 iterations. This time horizon was chosen because of the long generation times that typify most freshwater turtles. Carrying capacity (K) was set to the initial population size and the population was considered extinct when there was only one individual remaining (Supporting information, Table S1). A detailed description of how population sizes were estimated can be found in the Supporting information, Methods S1.

To estimate the influence of bycatch mortality, the harvest function in Vortex was used to incorporate different levels of annual mortality. Since male mortality in iteroparous species has little influence on the population size (Brooks *et al.,* 1991), only female mortality was modelled. Mortality rates that were modelled ranged from 1–5 adult females annually. An extreme case was also modelled where 10 adult females were removed annually. For each mortality rate, the final population size after 500 years and the probability of extirpation of the species from Lake Opinicon during the same timeframe were evaluated.

The influence of BRDs and BRSs on population persistence was not directly modelled because they would simply act to decrease mortality, which in the PVAs is equivalent to decreasing the harvest rate. For example, if BRDs are 90% effective for one species, then comparing a harvest rate of one individual and a harvest rate of 10 individuals is the same as comparing the impact of bycatch on a population with and without a BRD, respectively.

Bycatch reduction programmes

A bycatch reduction programme was considered consisting of two approaches: gear modifications (BRDs) and effort reduction (BRSs), which can be used independently or in combination (Hall *et al.,* 2000). For this study, one example of each was used; however, a summary table of other options that are available can be found in Supporting information, Table S2.

Exclusion devices

The devices used in this study were vertically oriented constriction BRDs affixed to the first funnel of the nets (Larocque *et al.,* 2012a). These devices work by physically preventing turtles from entering the net while allowing entry of target fish, using minimum diameter (carapace height; CH) as a selective criterion (Broadhurst, 2000; Roosenburg and Green, 2000). In each species, the number of females that would be excluded based on their CH by two devices with gap widths of 50 and 80 mm were considered. In musk turtles, the results from field trials using an 80 mm exclusion device (Larocque *et al.,* 2012a) were also included. Using calipers, maximum carapace height was measured to the nearest millimetre (Mosimann and Bider, 1960) of turtles captured in Lake Opinicon in 2011 (N = 750) and 2012 (N = 261) to determine the proportion of females that would be excluded for each width of BRD (Table 1).

Season length

At present the Lake Opinicon fishery is open from 1 January until 20 June and then it reopens the first Monday in September until 31 December. A proposed reduction in the season would be to close the fishery a month earlier on 20 May. Based on the mark–recapture sampling that occurred between late-April and mid-September in 2011 and 2012, the mean number of each species of turtle (both sexes) captured in each 24 h net set (expressed as catch per unit effort (CPUE)) in three time periods, late-April to 20 May, late-April to 20 June, and 21 May to 20 June was compared using an ANOVA (SAS, 2002). A post hoc Tukey HSD analysis was used to determine the origin of the differences when the ANOVA was significant.

Table 1. Mean carapace height (CH) for turtles captured in 2011 and 2012 in Lake Opinicon, Ontario, Canada. Turtles with a CH less than 51 mm were assumed to be able to pass through both the 50 mm and the 80 mm exclusion devices. Turtles with CH less than 81 mm were assumed to be able to pass through the 80 mm device and all turtles with a CH of 81 mm or larger were assumed to be excluded. Results from Larocque *et al.* (2012a) were included because of their success at musk turtle exclusion

Species	No. of individuals	Mean carapace height (mm)	Excluded in 50 mm device (%)	Excluded in 80 mm device (%)	Excluded in Larocque *et al.*, 2012 (%)
Painted	121	56.9 ± 5.1	92	0	—
Musk	134	46.8 ± 5.5	27	0	77
Snapping	22	124.0 ± 19.2	100	95	—
Map	141	78.1 ± 14.4	97	52	—

RESULTS

Population viability analysis

In all four species, any increase in annual adult mortality led to a decrease in population size over 500 years (Supporting information, Table S3; Figure 1). The magnitude of this decrease and the probability of extirpation from Lake Opinicon over 500 years were variable among species. Musk turtles had the largest population estimates in Lake Opinicon. Minimal changes in population size were noted with mortality from 1–5 adult females (a decrease of 1–5%). Extirpation of the population occurred within 350 years when 10 adult females were removed (Figure 1). An increase in mortality greater than two adult females per year was sufficient to extirpate the population of painted turtles from Lake Opinicon within 350 years (Figure 2). Mortality greater than 10 was sufficient to extirpate the population within 75 years. Similarly, the map turtle population showed a 98% decline with removal of one female annually. Extirpation within 300 years occurred for all higher rates of mortality, occurring in less than 50 years when annual mortality peaked at 10 individuals (Figure 2). In addition to having the lowest population size estimates, snapping turtles were also the most susceptible to removal with comparatively rapid extirpation (within 200 years) for all rates of additional annual adult female mortality considered.

Bycatch reduction programmes

Exclusion devices

Based on turtle carapace height, the 50 mm exclusion device would be expected to exclude 100% of

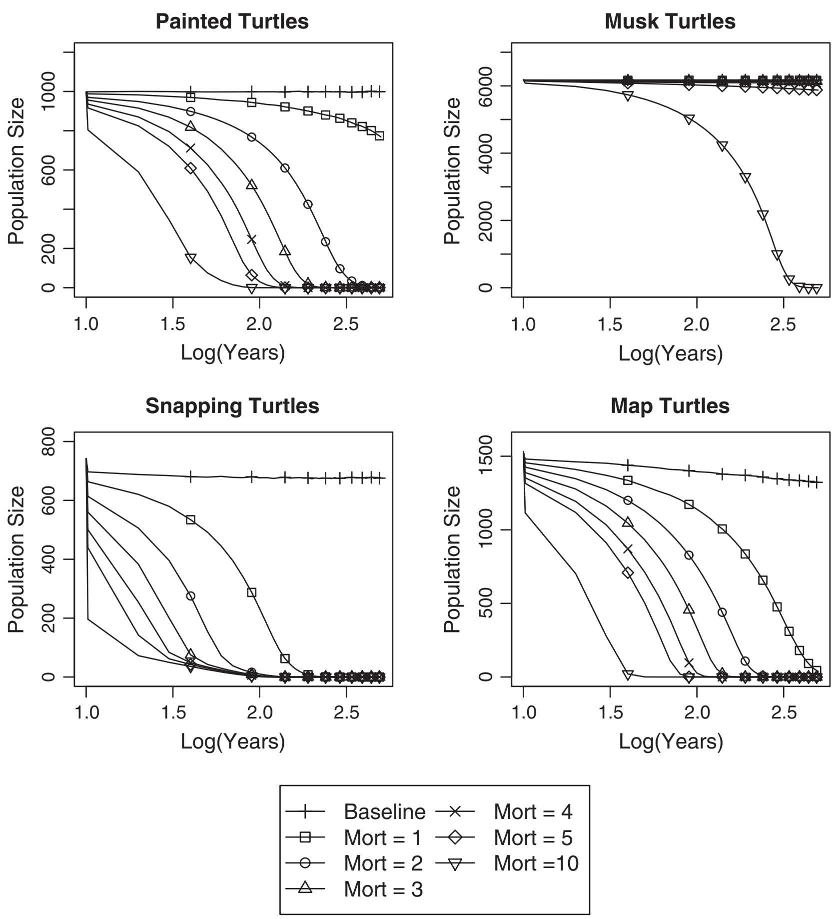

Figure 1. Impact of bycatch mortality on mean turtle population sizes of Lake Opinicon, Ontario, Canada based on a population viability analysis over 500 years. The x-axis has been log transformed to emphasize changes over the first 300 years.

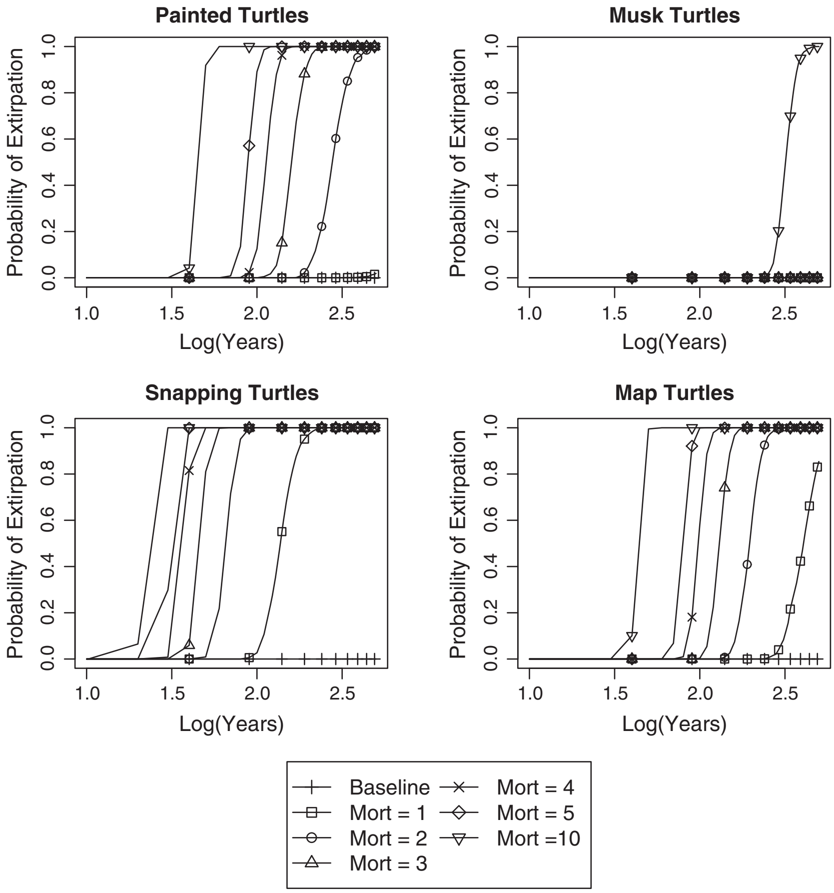

Figure 2. Impact of bycatch mortality on the mean probability of extirpation for freshwater turtle populations in Lake Opinicon, Ontario, Canada based on a population viability analysis over 500 years. The x-axis has been log transformed to emphasize changes over the first 300 years.

snapping turtles, as well as the majority of both map (97%) and painted (92%) turtles; however, only 27% of female musk turtles would be excluded (Figure 3). The larger opening on the 80 mm exclusion device would still exclude a majority of female snapping turtles (95%), but the proportion of turtles excluded decreased in map turtles (52%) and would probably not prevent access of any female painted or musk turtles (Figure 3).

Season length

The assessment of changes in seasonal capture indicated that despite variable catch throughout the spring and early summer, significantly more painted turtles were captured before 20 May than between 20 May and 20 June (Figure 4; ANOVA, $F = 3.478$, $P = 0.0345$, $DF_{Model} = 2$, $DF_{Error} = 103$). In contrast, significantly fewer snapping turtles

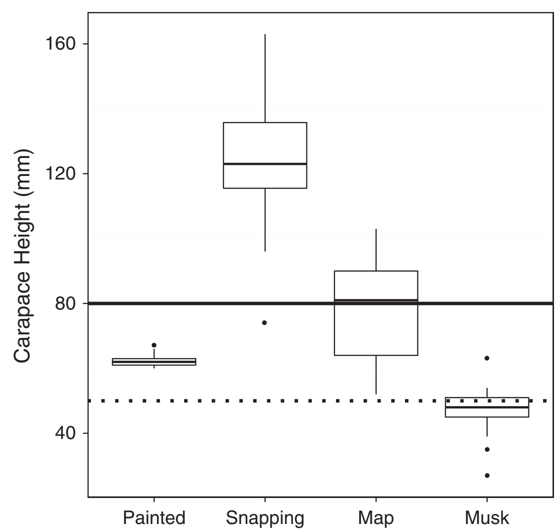

Figure 3. Mean and range of carapace heights for female painted, musk, snapping, and map turtles compared with the openings of the 50 mm and 80 mm bycatch exclusion devices.

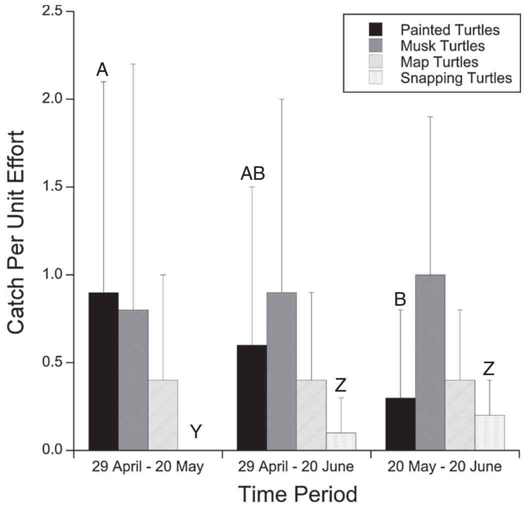

Figure 4. Catch per unit effort (CPUE) for turtles during 24 h fyke-net sets in Lake Opinicon, Ontario, Canada; all numbers are expressed as the mean number of turtles per net during the different time periods. Different letters denote significantly different values of CPUE for each species.

were captured before 20 May (ANOVA, $F = 8.717$, $P < 0.001$, $DF_{Model} = 2$, $DF_{Error} = 103$), with a peak in capture occurring in mid-June. There were no significant differences in the rate of capture of musk (ANOVA, $F = 0.274$, $P = 0.761$, $DF_{Model} = 2$, $DF_{Error} = 103$) or map turtles (Figure 4; ANOVA, $F = 0.012$, $P = 0.988$, $DF_{Model} = 2$, $DF_{Error} = 103$).

DISCUSSION

Although studies have documented additional mortality and resulting population effects in several animal taxa, this study is, to the best of our knowledge, the first attempt to link a PVA with two bycatch reduction strategies to estimate the long-term effects of bycatch-related mortality on a community of freshwater turtles. The first goal was to determine whether regular fishing pressure (and the potential bycatch of adult turtles) might have population-level effects in four species of freshwater turtles. In all four species, adult female mortality as a result of bycatch had an adverse impact on the population. This was most pronounced in snapping turtles, where an annual additional mortality rate as low as one female was predicted to reduce the population to zero within 200 years. Although the other three species appeared slightly more resilient to bycatch mortality, their increased resilience may be a result of higher initial population sizes as well as species-specific differences in age at maturity, clutch size and frequency, and the proportion of females that reproduce each year. For example, musk turtles appeared to be the most resilient to bycatch, but they also had the largest initial population size, lowest age of female maturity, and highest proportion of females reproducing each year. All of these demographic traits increase the annual reproductive capacity of the musk turtle population and allow it to better compensate for increased adult female mortality. Overall, despite differences in life-history parameters between the four species, proportionately small levels of adult female mortality as a result of bycatch could lead to their local extirpation.

Even low levels of additional adult mortality in iteroparous species like freshwater turtles can have serious effects on population sizes (Brooks *et al.,* 1991; Congdon *et al.,* 1994). These findings are consistent with those of Bulté *et al.* (2009) who predicted that low levels of adult female mortality caused by boat collisions could cause extirpation of map turtles in two populations, one of which was in Lake Opinicon. Several threats to freshwater turtles are well documented (e.g. habitat loss); however, threats such as fisheries bycatch are less apparent, but should not be overlooked. Although this case study was carried out in Lake Opinicon, Ontario, Canada, the results should be applicable to other instances and regions where turtles are captured as bycatch in commercial fishing nets. Furthermore, while the life-history parameters used in the models are specific to northern turtles, adjustments to some of these parameters for turtles in more southerly climates (i.e. faster growth rates, earlier age at maturity) may show that southern populations are more buffered against levels of bycatch mortality similar to those observed in this study. We caution, however, that in more southerly regions warmer temperatures may not only alter the life history of turtles (Iverson, 1992), but may also allow longer fishing seasons and increased net mortality owing to warmer water temperatures and the resulting shorter submergence times that turtles can tolerate (Fratto *et al.,* 2008a).

Perhaps the most important contribution of this study is the finding that simple BRPs may improve the probability of persistence of imperilled freshwater turtle populations. Exclusion BRDs have the potential to be highly effective at preventing capture (and ultimately mortality) in all four species of turtles, but especially in snapping turtles, a species that was particularly sensitive to additional adult mortality. As carapace height (CH) was the selective criterion used by constriction BRDs, the smaller device (50 mm) should keep out female map and painted turtles; however, the larger device (80 mm) would allow females of most species to enter the net. The numbers of turtles excluded presented in this study are minimum estimates because they are based on whether a turtle's carapace could physically pass through the device. Actual reduction rates achieved with constriction BRDs are higher than the values presented (Larocque *et al.,* 2012a; Cairns *et al.*, 2013). For example, in field trials the 80-mm exclusion device excluded 73% of musk turtles (Larocque *et al.,* 2012a), which may reflect behavioural avoidance as 80 mm is substantially larger than the CH of most musk turtles (Cairns *et al*., 2013).

Shortening the fishing season, in the case of Lake Opinicon shifting the closing date from 20 June to 20 May, is a good strategy to limit turtle bycatch (based on peak activity) and also to reduce mortality rates of those turtles that do enter the nets (based on the increased ability of turtles to withstand submergence in cool water; Ultsch, 1985). While seasonal activity rates vary among species, a shortened fishing season would still decrease the total number of turtles captured, provided that there is no compensatory increase in fishing effort. An increase in fishing effort could have adverse impacts on all turtles, but especially on painted turtles that are more active in the early spring. Appropriate changes to fishing season lengths would have to be determined for each geographic area because turtle activity is tied more closely to water temperature than to seasonality per se.

An important consideration when making recommendations on potential BRPs is the feasibility of their implementation. This issue has previously been reviewed (Campbell and Cornwell, 2008) with a clear need for more research directed at identifying the best manner with which to garner support from fishers. A recent study of the perspective of fishers involved in the eastern Ontario small-scale freshwater fishery suggested that there is generally a lack of recognition of turtle bycatch as a conservation issue (Nguyen *et al.,* 2013). Specifically, fishers did not perceive turtle bycatch as a significant threat to turtle populations and therefore did not understand the need for BRPs. The present study helps demonstrate the potential impact that even low bycatch mortality can have on regional turtle populations and may help convince fishers of the importance of BRP adoption.

At present, the conservation measures evaluated in this study have not been implemented within the province of Ontario. Of the two strategies proposed, changes to the fishing season may be more challenging to implement and, with the exception of musk turtles, may not be as effective as exclusion from the nets. Nguyen *et al.* (2013) reported that some fishers had voluntarily made changes to their nets (providing access to air) or the setting location (depth) to avoid turtle bycatch, but general adoption may require financial incentives. As an alternative, Nguyen *et al.* (2013) recommended the adoption of a 'co-management' approach; sharing responsibility among government, fishers, and other stakeholders. In addition, other BRSs and BRDs are available (summarized in Supporting information, Table S2), but were not evaluated specifically here. BRPs that reduce mortality (i.e. harvest season, exclusion devices, air pockets) can be applied either independently or in tandem to help alleviate population effects associated with bycatch mortality (Hall *et al.,* 2000). A combination of BRSs and BRDs could help nearly eliminate mortality, which is essential for the persistence of freshwater turtle populations in areas where commercial fishing occurs.

One important caveat regarding the population modelling in this study is that it was assumed that the only additional mortality in the population (beyond natural mortality levels) was from bycatch. Bycatch is only one of several additional sources of mortality in freshwater turtles, including road mortality (Gibbons, 1970), environmental toxins (Bell *et al.*, 2006; Van Meter *et al.*, 2006), boat collisions (Bulté *et al.,* 2009), and recreational harvest (Congdon *et al.,* 1994; Gamble and Simons, 2004). The compound effect of these other sources of mortality would further decrease population size, and thus exacerbate the impact of bycatch mortality on population size and population persistence. Therefore, the probabilities of persistence presented for the four species of freshwater turtles in Lake Opinicon are optimistic scenarios and should be considered as such.

Turtles captured as bycatch in commercial fishing nets often perish. The results of this study demonstrate that as long-lived, iteroparous species, freshwater turtles are highly sensitive even to low levels of additional adult mortality. Furthermore, population effects (i.e. halving of the population) in many species are unlikely to be apparent for 25–50 years. These relatively slow population declines are difficult to detect on a human-life timescale and could make it challenging to convince stakeholders that local turtle populations are indeed in decline. Nevertheless, it is imperative that appropriate bycatch mitigation measures, including BRPs, are put in place to ensure the long-term persistence of freshwater turtles.

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REFERENCES

Agbeti MD, Kingston JC, Smol JP, Watters C. 1997. Comparison of phytoplankton succession in two lakes of different mixing regimes. *Archives of Hydrobiology* **140**: 37–69.

Allan JD, Abell R, Hogan ZEB, Revenga C, Taylor BW, Welcomme RL, Winemiller K. 2005. Overfishing of inland waters. *BioScience* **55**: 1041–1051.

Barko VA, Briggler JT, Ostendorf DE. 2004. Passive fishing techniques: a cause of turtle mortality in the Mississippi River. *Journal of Wildlife Management* **68**: 1145–1150.

Beissinger S. 2002. Population viability analysis: past, present, future. In *Population Viability Analysis*, Beissinger S, McCullough DR (eds). University of Chicago Press: Chicago; 5–17.

Bell B, Spotila JR, Congdon J. 2006. High incidence of deformity in aquatic turtles in the John Heinz National Wildlife Refuge. *Environmental Pollution* **142**: 457–465.

Bishop JM. 1983. Incidental capture of diamondback terrapin by crab pots. *Estuaries* **6**: 426–430.

Boyce MS. 1992. Population viability analysis. *Annual Review of Ecology and Systematics* **23**: 481–506.

Broadhurst MK. 2000. Modifications to reduce bycatch in prawn trawls: a review and framework for development. *Reviews in Fish Biology and Fisheries* **10**: 27–60.

Brook BW, O'Grady JJ, Chapman AP, Burgman MA, Akcakaya HR, Frankham R. 2000. Predictive accuracy of population viability analysis in conservation biology. *Nature* **404**: 385–387.

Brooks RJ, Brown GP, Galbraith DA. 1991. Effects of a sudden increase in natural mortality of adults on a population of the common snapping turtle (*Chelydra serpentina*). *Canadian Journal of Zoology* **69**: 1314–1320.

Bulté G, Carrière MA, Blouin-Demers G. 2009. Impact of recreational power boating on two populations of northern map turtles (*Graptemys geographica*). *Aquatic Conservation: Marine and Freshwater Ecosystems* **20**: 31–38.

Burns C. 2007. Biological sustainability of commercial fishing in the inland waters of Kemptville District. Ontario Ministry of Natural Resources Technical Paper, Kemptville, ON.

Bury RB. 2011. Modifications of traps to reduce bycatch of freshwater turtles. *Journal of Wildlife Management* **75**: 3–5.

Cairns NA, Stoot LJ, Blouin-Demers G, Cooke SJ. 2013. Refinement of bycatch reduction devices to exclude freshwater turtles from commercial fishing nets. *Endangered Species Research* **22**: 251–261.

Campbell LM, Cornwell ML. 2008. Human dimensions of bycatch reduction technology: current assumptions and directions for future research. *Endangered Species Research* **5**: 325–334.

Congdon JD, Greene JL, Gibbons JW. 1986. Biomass of freshwater turtles: a geographic comparison. *American Midland Naturalist* **115**: 165–173.

Congdon JD, Dunham AE, Van Loben Sels RC. 1994. Demographics of common snapping turtle (*Chelydra serpentina*): implications for conservation and management of long-lived organisms. *American Zoologist* **34**: 397–408.

Coulson T, Mace GM, Hudson E, Possingham H. 2001. The use and abuse of population viability analysis. *Trends in Ecology and Evolution* **16**: 219–221.

Crowder LB, Murawski SA. 1998. Fisheries bycatch: implications for management. *Fisheries* **23**: 8–17.

Dorcas ME, Willson JD, Gibbons JW. 2007. Crab trapping causes population decline and demographic changes in diamondback terrapins over two decades. *Biological Conservation* **137**: 334–340.

Fratto ZW, Barko VA, Pitts PR, Sheriff SL, Briggler JT, Sullivan KP, McKeage BL, Johnson TR. 2008a. Evaluation of turtle exclusion and escapement devices for hoop-nets. *Journal of Wildlife Management* **72**: 1628–1633.

Fratto ZW, Barko VA, Scheibe JS. 2008b. Development and efficacy of a bycatch reduction device for Wisconsin-type fyke nets deployed in freshwater systems. *Chelonian Conservation Biology* **7**: 205–212.

Galbraith DA. 1986. Age estimates, survival, growth, and maturity of female *Chelydra serpentina* Linnaeus in Algonquin Provincial Park, Ontario. MSc thesis, University of Guelph, Canada.

Gamble T, Simons AM. 2004. Comparison of harvested and nonharvested painted turtle populations. *Wildlife Society Bulletin* **32**: 1269–1277.
Gibbons JW. 1970. Sex ratios in turtles. *Researches on Population Ecology* **12**: 252–254.
Goldsworthy SD, Page B. 2007. A risk-assessment approach to evaluating the significance of seal bycatch in two Australian fisheries. *Biological Conservation* **139**: 269–285.
Hall MA, Alverson DL, Metuzals KI. 2000. By-catch: problems and solutions. *Marine Pollution Bulletin* **41**: 204–219.
Harwood J. 2000. Risk assessment and decision analysis in conservation. *Biological Conservation* **95**: 219–226.
Iverson JB. 1991. Patterns of survivorship in turtles (order Testudines). *Canadian Journal of Zoology* **69**: 385–391.
Iverson JB. 1992. Correlates of reproductive output in turtles (Order Testudines). *Herpetological Monographs* **6**: 25–42.
Kennelly SJ, Broadhurst MK. 2002. By-catch begone: changes in the philosophy of fishing technology. *Fish and Fisheries* **3**: 340–355.
Lacy RC. 1993. VORTEX: a computer simulation model for population viability analysis. *Wildlife Research* **20**: 45–65.
Lacy RC. 2000. Structure of the VORTEX simulation model for population viability analysis. *Ecological Bulletins* **48**: 191–203.
Larocque SM, Cooke SJ, Blouin-Demers G. 2012a. Mitigating bycatch of freshwater turtles in passively fished fyke nets through the use of exclusion and escape modifications. *Fisheries Research* **125**: 149–155.
Larocque SM, Colotelo AH, Cooke SJ, Blouin-Demers G, Haxton T, Smokorowski KE. 2012b. Seasonal patterns in bycatch composition and mortality associated with a freshwater hoop net fishery. *Animal Conservation* **15**: 53–60.
Lowry MB, Pease BC, Graham K, Walford TR. 2004. Reducing the mortality of freshwater turtles in commercial fish traps. *Aquatic Conservation: Marine and Freshwater Ecosystems* **15**: 7–21.
Majluf P, Babcock EA, Riveros JC, Schreiber MA, Alderete W. 2002. Catch and bycatch of sea birds and marine mammals in the small-scale fishery of Punta San Juan, Peru. *Conservation Biology* **16**: 1333–1343.
McCarthy MA, Andelman SJ, Possingham HP. 2003. Reliability of relative predictions in population viability analysis. *Conservation Biology* **17**: 982–989.
Melvin EF, Parrish JK, Conquest LL. 2001. Novel tools to reduce seabird bycatch in coastal gillnet fisheries. *Conservation Biology* **13**: 1386–1397.
Michaletz PH, Sullivan KP. 2002. Sampling channel catfish with tandem hoop nets in small impoundments. *North American Journal of Fisheries Management* **22**: 870–878.
Morris WF, Doak DF. 2002. *Quantitative Conservation Biology: Theory and Practice of Population Viability Analysis.* Sinauer Associates: Sunderland, MA, USA.
Mosimann JE, Bider JR. 1960. Variation, sexual dimorphism, and maturity in a Quebec population of the common snapping turtle, *Chelydra serpentina*. *Canadian Journal of Zoology* **38**: 19–38.
Nguyen VM, Larocque SM, Stoot LJ, Cairns NA, Blouin-Demers G, Cooke SJ. 2013. Perspectives of fishers on turtle bycatch and conservation strategies in a small-scale inland commercial fyke net fishery. *Endangered Species Research* **22**: 11–22.
Ontario Ministry of Natural Resources (OMNR). 2013. Ontario Commercial Fishing License, License Appendix B - Conditions. Lake Ontario Management Unit, Picton, ON.
Raby GD, Colotelo AH, Cooke SJ, Blouin-Demers G. 2011. Freshwater commercial bycatch: an understated conservation problem. *BioScience* **61**: 271–280.
Roosenburg WM, Green JP. 2000. Impact of a bycatch reduction device on diamondback terrapin and blue crab capture in crab pots. *Ecological Applications* **10**: 882–889.
SAS. 2002. JMP version 9 statistics and graphics guide. SAS, Cary, NC.
Sullivan KP, Gale CM. 1999. A comparison of channel catfish catch rates, size distributions, and mortalities using three different gears in a Missouri impoundment. In *Catfish 2000: Proceedings of the International Ictalurid Symposium*, Irwin ER, Hubert WA, Rabeni CF, Schramm HL, Coon T (eds). American Fisheries Society Symposium 24: Bethesda; 293–300.
Taylor BL. 2002. The reliability of using population viability analysis for risk classification of species. *Conservation Biology* **9**: 551–558.
Turtle Taxonomy Working Group [van Dijk PP, Iverson JB, Shaffer HB, Bour R, Rhodin AGJ]. 2012. Turtles of the world, 2012 update: annotated checklist of taxonomy, synonymy, distribution, and conservation status. In *Conservation Biology of Freshwater Turtles and Tortoises: A Compilation Project of the IUCN/SSC Tortoise and Freshwater Turtle Specialist Group*, Rhodin AGJ, Pritchard PCH, van Dijk PP, Saumure RA, Buhlmann KA, Iverson JB, Mittermeier RA (eds). Chelonian Research Monographs No. 5; 000.243–000.328.
Ultsch GR. 1985. The viability of nearctic freshwater turtles submerged in anoxia and normoxia at 3 and 10 °C. *Comparative Biochemistry and Physiology. Part A, Physiology* **81**: 607–611.
Van Meter RJ, Spotila JR, Avery HW. 2006. Polycyclic aromatic hydrocarbons affect survival and development of common snapping turtle (*Chelydra serpentina*) embryos and hatchlings. *Environmental Pollution* **142**: 466–475.

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