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Golf courses and wetland fauna

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Abstract. Golf courses are often considered to be chemical-intensive ecosystems with negative impacts on fauna. Here we provide evidence that golf courses can contribute to the support and conservation of wetland fauna, i.e., amphibians and macroinvertebrates. Comparisons of amphibian occurrence, diversity of macroinvetebrates, and occurrence of species of conservation concern were made between permanent freshwater ponds surveyed on golf courses around Sweden's capital city, Stockholm, and off-course ponds in nature-protected areas and residential parklands. A total of 71 macroinvertebrate species were recorded in the field study, with no significant difference between golf course ponds and off-course ponds at the species, genus, or family levels. A within-group similarities test showed that golf course ponds have a more homogenous species composition than ponds in nature-protected areas and ponds in residential parkland. Within the macroinvertebrate group, a total of 11 species of odonates were identified, with no difference detected between the categories of ponds, nor any spatial autocorrelation. Significant differences were found between pond categories in the occurrence of five species of amphibians, although anuran occurrence did not differ between ponds. The great crested newt (*Triturus cristatus*) was significantly associated with golf course ponds, but the smooth newt (*Triturus vulgaris*) was not. We found no evidence of any correlation between pond size and occurrence of amphibians. Among the taxa of conservation concern included in the sample, all amphibians are nationally protected in Sweden, with the internationally threatened *T. cristatus* more frequently found in golf course ponds. Among macroinveterbrates of conservation status, the large white-faced darter dragonfly (*Leucorrhinia pectoralis*) was only detected in golf course ponds, and *Tricholeiochiton fagesi* (Trichoptera) was only found in one off-course pond. GIS results revealed that golf courses provide over a quarter of all available permanent, freshwater ponds in central greater Stockholm. We assert that golf courses have the potential to contribute to wetland fauna support, particularly in urban settings where they may significantly contribute to wetland creation. We propose a greater involvement of ecologists in the design of golf courses to further bolster this potential.

Key words: amphibians; biodiversity; conservation; ecosystem management; golf courses; land use; macroinvertebrates; odonates; ponds; wetlands.

Introduction

Considering that the great majority of threatened and endangered species occur on private lands, e.g., >90% in the United States (Scott et al. 2001), it is often suggested that such lands should be more closely integrated in biodiversity management schemes (Oldfield et al. 2003, Kiesecker et al. 2007). The Millennium Ecosystem Assessment (MA) and the Ecological Society of America (ESA) have also called for more initiatives to improve cooperation and forge partnerships among people and different sectors in society to bolster ecosystem management (MA 2005, Palmer et al. 2005). In this paper we examine the potential of the recreational land use of golf in nature conservation and management with a focus on wetland fauna.

Currently, there exist over 31 500 golf courses worldwide (Tanner and Gange 2005), with some geographical parts, e.g., Europe and the United States, having experienced rapid golf course development in recent decades. Europe holds ~5800 golf courses. Their establishment increased by an average of 5% per year between 1990 and 2000 (EIGCA 2007). The 17 000 or so golf courses in the United States cover >600 000 ha of land (Birchfield and Deters 2005), a land area larger than the state of Delaware and nearly half of Connecticut. The number of new courses constructed in the United States in the 1970s through the 1990s averaged >300 courses per year, i.e., nearly one new golf course per day was constructed during this 30-year period (Nicholls and Crompton 2007). Golf course development has also been burgeoning in parts of Australia (Hodgkison et al. 2007), Japan (Yasuda and Koike

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2006), and Southeast Asia, and is booming in China, where local governments believe golf courses attract investors.

Despite the large number of golf courses and the vast land area they occupy, few ecological studies on them exist in the scientific literature. High avifauna species richness, diversity, and abundance have been found on golf courses, including threatened and regionally declining bird populations (e.g., Cristol and Rodewald 2005, Merola-Zwartjes and DeLong 2005, Rodewald et al. 2005), and other species of conservation concern (Hodgkison et al. 2007), as well as essential pollinators, and predators of noxious insects (e.g., Blair and Launer 1997, Gange and Lindsay 2002). Golf courses also preserve endangered habitat types, such as dune grassland and inland heathland in Europe (Gange et al. 2003) and riparian vegetative communities and the eastern longleaf pine (*Pinus palustris*) ecosystems in the United States (Heuberger and Putz 2003, Merola-Zwartjes and DeLong 2005).

Considerably less attention has been given to the value of golf courses for aquatic-dependent biota, despite the fact that water bodies normally are incorporated on golf courses to provide hazard features. It has been estimated that a typical English golf course contains some 2% (or roughly 1.1 ha) of wetland cover (Dair and Schofield 1990). These wetlands consist of permanent lakes and ponds, streams, creeks, and estuaries, as well as seasonal wetlands (Scott et al. 2002). While a number of studies have evaluated the movement of fertilizers and pesticides from golf courses to groundwater and surface water (see, e.g., Ryals et al. 1998, Lewis et al. 2001), with a major review by Cohen et al. (1999), which concludes that there are generally no significant human toxicological impacts, little is known about golf courses' chemical impacts on wetland fauna (Winter et al. 2002). To our knowledge, this paper represents the first peer-reviewed, European assessment of amphibians and macroinvertebrates in golf course water bodies, with a focus on central greater Stockholm, the capital province of Sweden. In this region, chemical inorganic applications, i.e., fertilizers (primary macronutrients) and pesticides, are regularly used on arable land and pastures, but also intensively used for turf management on golf courses. In Swedish agricultural regions both private and public water sources may be contaminated from leakage of nitrogen (Jansson and Colding 2007), with censuses showing the presence of pesticides in wells from contamination of groundwater (SNV 1999*a*). It has been estimated that between 0.1% and 0.3% of pesticides associated with agriculture ultimately reach lakes and other bodies of water (SNV 1999*b*), with potentially harmful impacts on aquatic biota (Karlström 1995).

Inorganic chemicals are, however, considerably less used in Sweden on publicly managed parklands, and completely avoided in nature-protected areas. In Stockholm, for example, it is prohibited to use pesticides in the management of parklands for public access, with the exception of treatment of the invasive giant hogweed (*Heracleum mantegazzianum*) (Maria-Gamla stans stadsdelsförvaltning 2005). Moreover, fertilizers are sparsely used in park management.

This discrepancy in land management regimes allowed us to test whether permanent freshwater ponds on golf courses differ from those in off-course ponds, located in nature-protected areas and residential parklands, regarding amphibian occurrence and diversity of macroinvertebrates, including species of conservation concern (i.e., internationally red-listed and nationally protected species). We hypothesized that golf courses represent chemically stressed environments for pond-dependent fauna, and reasoned that a difference in fauna would be expressed between golf course ponds and off-course ponds due to difference in land management. In addition to the fauna inventory, we generate data on pond distribution in central greater Stockholm by way of a geographic information system (GIS).

Focal organism groups

Amphibians, and several macroinvertebrates, e.g., odonates (Anisoptera) and damselflies (Zygoptera), are in decline in many parts of the world (Houlahan et al. 2000, Carchini et al. 2005). Amphibians are generally more sensitive to environmental toxins and trophic disturbance than other vertebrate groups, due to their exposure to both terrestrial and aquatic habitats during their life cycles, as well as because they are relatively long lived, and highly philopatric (Welsh and Ollivier 1998, Alford and Richards 1999). Because amphibians have highly permeable skin, they may be particularly susceptible to both soil and water contaminants (Howard et al. 2002). There are many studies demonstrating harmful effects on amphibians from waters contaminated by pesticides and fertilizers (e.g., Watt and Oldham 1995, Raloff 1998). In addition to environmental acidity and toxicants, amphibian declines and losses have complex causes, e.g., ultraviolet radiation, predation, habitat modification, stochastic extinctions, alteration in climate and weather patterns, and interactions among these factors (Alford and Richards 1999, Marsh and Trenham 2001). In Europe, amphibian declines are primarily associated with habitat modification, including draining of wetlands, which increases the probability of regional extinctions (Alford and Richards 1999). Urbanization is also recognized as a key factor in the loss of bodies of water and the elimination of many amphibian populations (Rubbo and Kiesecker 2005), where the spread of highly developed land acts as barriers to dispersal between amphibian breeding habitats (Fahrig et al. 1995, Marsh and Trenham 2001). Semiaquatic species like amphibians breed and lay eggs in wetlands during short breeding seasons of a few days or weeks, and during the rest of the year migrate to terrestrial habitats to forage and overwinter (Semlitsch and Bodie 2003).

Sweden holds 13 recorded amphibian species, with five inhabiting the Stockholm region, including the common toad (*Bufo bufo*), the moor frog (*Rana arvalis*), the common frog (*R. temporaria*), the smooth newt (*Triturus vulgaris*), and the great crested newt (*T. cristatus*). Declines of these amphibians have been documented in Stockholm from time-series censuses. From 1992 to 1996, a significant decline was recorded for the common frog (*R. temporaria*), while the other species increased in the number of breeding sites examined (i.e., small lakes, seasonal wetlands, and permanent ponds). However, the total number of water bodies occupied by any amphibian species has decreased, with an associated increase of species co-occurrence in localities (Löfvenhaft et al. 2002).

Macroinvertebrates are regularly used for monitoring change and conditions in freshwaters in Sweden (e.g., Wiederholm and Johansson 1999) and elsewhere (e.g., Winter et al. 2002). Odonates (i.e., dragonflies) have been used more widely as bioindicators, especially larvae that often live several years at a site, and thus provide a means of ensuring continuity in sampling of water conditions for both running and still waters (Stewart and Samways 1998, Carchini et al. 2005, Foote and Rice Hornung 2005). For example, odonates have been used as indicator species for evaluating the habitat value of ponds (Carchini et al. 2005), and as a criterion for the selection of Sites of Special Scientific Interest (SSSIs) in the United Kingdom (Briers and Biggs 2003). Because odonates inhabit both aquatic and terrestrial habitats during their life cycle, they may better reflect disturbance to the transitional riparian buffer, as compared with strict wetland obligates (Foote and Rice Hornung 2005).

With a distribution continuum from temporary to permanent waters, Europe holds 164 species of odonates, out of which 37% are considered to be threatened, and with populations as a whole presently declining (McLean et al. 1999, Carchini et al. 2005). Habitat fragmentation is a primary cause of this decline, resulting in nonrandom population extinctions (Purse et al. 2003). Sweden holds a total of 61 species of odonates, with five nationally red-listed and six internationally protected, according to the European Habitat Directive, Annex II and IV (Dannelid et al. 2008). In the most recent survey of greater Stockholm, a total of 38 species of odonates were recorded (Ekestubbe et al. 2003).

Methods

Study area characteristics

Of Europe's >6000 golf courses, Sweden holds 499 courses, covering 25 500 ha of land (SCB 2006, SGF 2007). Sweden is the fifth most golf-course-rich country in Europe. The great majority of these courses are confined to urban areas, representing privately managed lands that sometimes are owned by local clubs. Greater Stockholm constitutes some 101 100 ha of land, and has 24 golf courses, making up 1.6% of the land area. These courses are scattered within a mosaic composed of urban built-up land, publicly managed parklands, forested land, and a tiny portion of arable land (Colding et al. 2006). A median-sized golf course in this area covers 57 ha, with 70% representing nonplayable areas that comprise smaller hillside patches, wetlands, stream banks, grasslands, groves, and woodlands (Colding et al. 2006). Hence, some 40 ha of a typical golf course consist of natural habitats. Considering that a Swedish median-sized nature reserve makes up some 20 ha of land (Nilsson and Götmark 1992), golf courses represent quite large, seminatural ecosystems. These golf courses also contain a considerable amount of water bodies (e.g., ditches, creeks, and ponds), with some courses holding between 10 and 20 ponds (Colding et al. 2006). Fauna surveillance of this study was restricted to a 50 700-ha study area, referred to here as central greater Stockholm (Fig. 1).

Study design and criteria for analyses

Ponds of interest in this study were those that can be defined as permanent, lentic water bodies (both man-made and natural), between 25 m^2 and 2 ha in area, following the definition used by Collinson et al (1995). A total of 24 ponds were selected for analysis of fauna: 12 golf course ponds (GPs) and 12 off-course ponds (OPs). For a map of pond distribution, see Fig. 1. A random sample (without replacement) of ponds was surveyed at the six most centrally located 18-hole golf courses around Stockholm city; hence, these courses could be characterized as highly urban impacted. They were all constructed in the years 1926 to 1987, with four created before 1933. One of these courses held only one pond that was searchable (the other was under reconstruction during our sampling period), with remaining courses having two or more ponds (range = 2–13 ponds), where two ponds per course were surveyed. To obtain our preset sample size of 12 golf course ponds, three ponds were therefore surveyed at the course having the greatest number of ponds.

For selection of off-course ponds we used a high-resolution digital map (1:10.000) of real estates, available from the National Land Survey of Sweden (Lantmäteriverket 2005). The 12 off-course ponds (OPs) were deliberately selected for surveillance of fauna. To limit the influence of geographic differences, each off-course pond was selected based on the closest distance from a given, surveyed golf course, and its location in publicly managed parkland or nature-protected area. Based on these criteria, seven off-course ponds were chosen for fauna surveillance in parklands, and five in nature-protected areas.

A GIS assessment was also carried out in the 50 700-ha study area, to determine the number, area, and proportion of permanent, lentic freshwater ponds, using ArcView v.3.2 GIS software (ESRI, Redlands, California, USA). For construction of the pond GIS layer, the

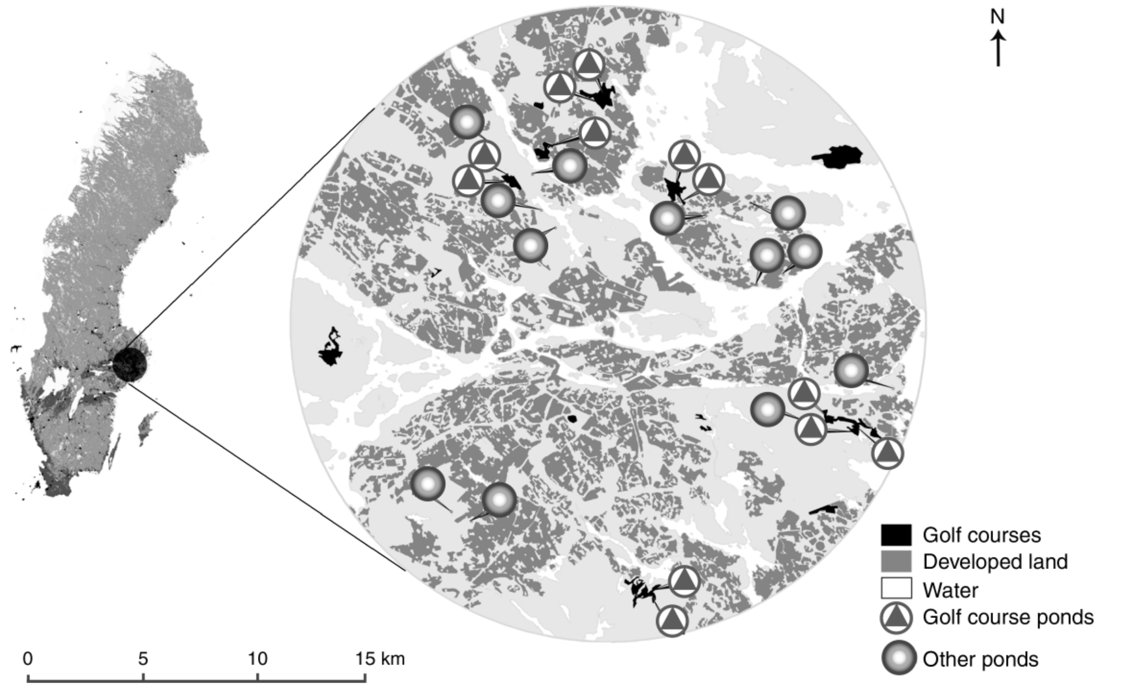

FIG. 1. Study area and distribution of 24 surveyed ponds in central greater Stockholm, Sweden.

following sources were used: a high-resolution digital map (1:10.000) available from the National Land Survey of Sweden (Lantmäteriverket 2005); the 1998 Stockholm biotope database; local orienteering maps of forest areas where ponds were harder to identify, due to tree shadowing; course maps of local golf courses; and, black-and-white digital aerial photos from a database provided by the National Land Survey (*available online*).[6]

Fauna survey

Fauna was sampled during spring and early summer of 2005 (4 May to 7 June) with sites positioned by GPS for GIS compatibility. At each site we determined pond size and the closest distance between a pond and any forest and/or shrub patch to determine potential terrestrial habitats of recorded amphibians. Terrestrial patches ranged from smaller shrub and tree patches to rather large forested areas with a mixture of native deciduous and coniferous trees.

Macroinvertebrates were sampled using a standard water net (mesh size 0.5×0.5 mm) in a 50 cm wide section outwards from the shoreline among the aquatic vegetation. An area of 9–25 m^2 was sampled with kick-sampling technique at each pond, depending on the size and structure of the ponds, assumed to represent the overall macroinvertebrate fauna associated with the different pond substrates (Sahlén and Ekestubbe 2001). The daytime sampling was carried out by the same person with an effort to achieve random sampling. Each pond was visited once with collected fauna assemblages preserved in 80% alcohol in the field, and identified at the laboratory to the level of individual taxa, i.e., species, genera, or families.

Amphibians were detected between dusk and midnight using visual encounter surveys (VES) by way of torchlight [using flashlights] (Campbell and Christman 1982). We counted the number of different adult amphibian species by walking around the pond, scanning every meter or so as gaps in pond bank vegetation allowed. Smaller ponds (<200 m^2) allowed for the whole water column to be surveyed by torching [shining flashlights]. At larger ponds, circumscribed banks (approximately ≤ 1 m water depth) were surveyed where vegetation gaps occurred. An effort was also made to encounter adult amphibians present in the vicinity of the ponds (approximately ≤ 50 m). To reduce possible influence from weather, we strived to sample amphibians in golf course ponds and off-course ponds that were located within the same geographic cluster during a given sampling day. Because amphibian behavior and site attributes vary seasonally and temporally with weather and internal rhythms of the animals (Alford and Richards 1999), sites with no observations of amphibians were revisited once to assure that they did not contain amphibians.

Statistical analyses

Differences in species composition of macroinvertebrates and amphibians were tested statistically using one-way analysis of similarities (ANOSIM) randomiza-

[6] ⟨https://geoimager.lantmateriet.se/digibib/⟩

tion test (Clark 1988), a nonparametric analogue to the standard univariate one- and two-way ANOVA tests. Analyses were done with the PRIMER v.6 software (Clarke and Gorley 2006).

To exclude potential deviations caused by the high number of individual taxa in some ponds, two separate runs were performed: one untransformed test (giving more weight to dominant species), and one on presence/absence. Due to difficulties with the identification of some organisms, tests for macroinvertebrates were run three times on different taxonomic levels: species, genera, and family. Since we had two types of off-course ponds, we also ran a set of analyses where ponds were classified in three categories of land use: golf courses, nature-protected areas, and residential parklands. Within-group similarities were checked for the three different pond categories. We also ran a more specific test on the species within the order Odonata. Within the amphibian group, separate tests were run for great crested and smooth newts (Mann-Whitney U test), respectively, and one for anurans (ANOSIM). A version of Mantel's test designed to test matched similarity matrices (called RELATE in PRIMER) was used to check for potential spatial autocorrelations. We also controlled for the different pond sizes by running a correlation analysis between pond size and the occurrence of the different amphibians.

Results

A total of 71 macroinvertebrate species were recorded in the field study. For macroinvertebrates there was no significant difference between golf course ponds and off-course ponds at any taxonomic level (species, genera, family), represented here by the species level, since it contains most data (untransformed, $R = 0.04$, $P = 0.167$; presence/absence, $R = 0.016$, $P = 0.352$). No statistically significant evidence of spatial autocorrelation was found (Rho $= 0.064$, $P = 0.243$). We found some weaker support for differences between ponds when we divided them into three categories (Untransformed, $R = 0.16$, $P = 0.051$; Presence/Absence, $R = 0.135$, $P = 0.056$), where the only significant pairwise difference was for presence/absence transformed data from golf course and residential parklands (Presence/Absence, $R = 0.234$, $P = 0.018$). Within-group similarities test showed that golf course ponds have a more homogenous species composition (average similarity: 32.27), compared with ponds in nature-protected areas (average similarity: 18.27) and ponds in residential parkland (average similarity: 14.89).

Within the macroinvertebrate group a total of 11 species of odonates were identified (Fig. 2), with 6 recorded in golf course ponds and 8 recorded in off-course ponds. However, no difference could be detected between the pond categories (Untransformed, $R = 0.001$, $P = 0.408$; Presence/Absence, $R = -0.001$; $P = 0.44$), nor any spatial autocorrelation (Rho $= 0.065$, $P = 0.206$).

In total we recorded five species of amphibians (Fig. 3). We obtained a significant difference for amphibians between golf course ponds and off-course ponds (Untransformed, $R = 0.129$, $P = 0.012$; Presence/Absence, $R = 0.137$, $P = 0.009$) with no statistical indication of spatial autocorrelation (Rho $= 0.083$, $P = 0.188$). Anurans did not differ between ponds (Untransformed, $R = 0.019$, $P = 0.186$; Presence/Absence, $R = 0.025$, $P = 0.199$). Nonparametric independent samples comparison by Mann-Whitney U test revealed that the great crested newt was significantly associated with golf course ponds ($Z = 2.05$, $P = 0.04$) while the smooth newt was not ($Z = 1.41$, $P = 0.157$). We found no evidence of any correlation between pond size and occurrence of amphibians.

Among taxa of conservation concern, two represent internationally red-listed species, i.e., the great crested newt and the large white-faced darter dragonfly (*Leucorrhinia pectoralis*), registered as "near threatened" in Appendix II of the Bern Convention, with the latter species only detected in golf course ponds. Moreover, *Tricholeiochiton fagesi* (Trichoptera), found in one off-course pond, is nationally red-listed (Gärdenfors 2005). In addition, all amphibians encountered in this study are nationally protected.

GIS results revealed that 167 freshwater ponds were confined to the study area, of which 44 (i.e., 26.3%) represent golf course ponds. This corresponds to well over twice the number of ponds found in protected areas, with golf course ponds making up a greater total pond area than ponds confined to nature-protected areas (Table 1). Ponds used in surveillance of fauna ranged in size from 50 m^2 to 10.350 m^2 (Table 2), with a mean size of 769 m^2 and 3104 m^2 for golf course ponds and off-course ponds, respectively. The measured distance between any pond and its nearest forest patch is given in Table 2.

Discussion

Golf course ponds as habitats for macroinvertebrates and amphibians

Our hypothesis that golf course ponds in central greater Stockholm represent chemically stressed habitats of little value for wetland fauna is not supported by the field data. We found no evidence that golf course ponds differ in providing habitats for macroinvertebrates (at any taxonomic level) relative to other types of ponds examined in this study. This relationship was true regardless of where golf courses were located in the study area. Interestingly, we found no significant difference in species composition between golf course ponds and ponds located in nature-protected areas. The only significant difference found was between golf course ponds and ponds located in residential parklands.

Two odonate species were only recorded in golf course ponds, including the large white-faced darter dragonfly (*Leucorrhinia pectoralis*) and the common blue damselfly (*Enallagma cyathigerum*), with the former

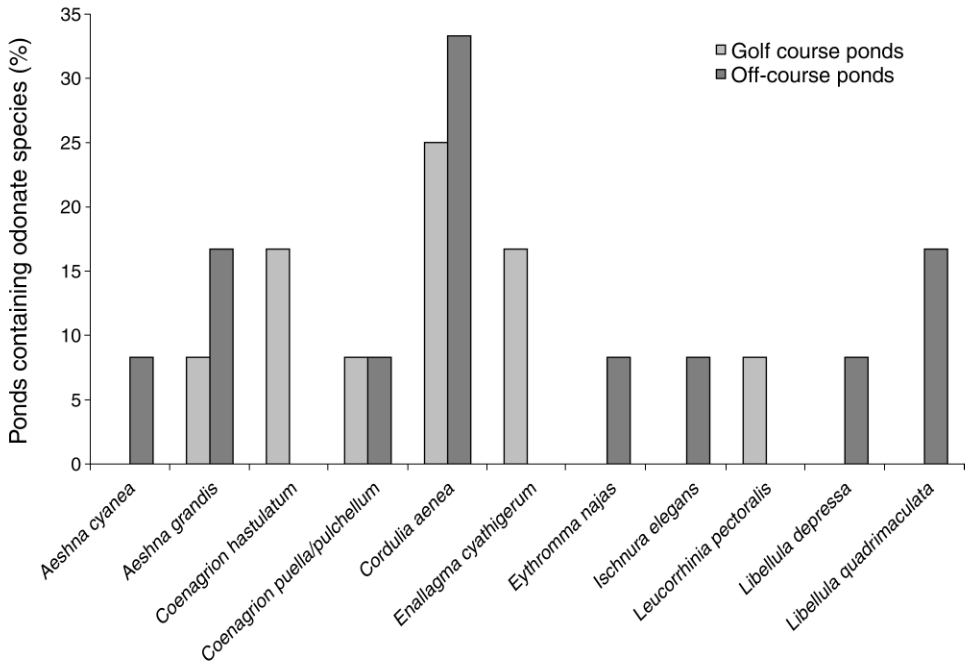

FIG. 2. Species of Odonata recorded in the case study, by pond location on or off golf courses.

being internationally red-listed. The presence of these species correlates with aquatic plant richness in Sweden (Sahlén and Ekestubbe 2001).

A significant difference for amphibians between pond categories was obtained. Anurans did not differ; however, the study reveals that golf course ponds are suitable habitats for newts. Both of the newt species in Sweden occurred in golf course ponds, with the great crested newt significantly associated with these ponds. This species strongly depends on water bodies with low levels of pollutants and acidification (Karlström 1995, Marsh and Trenham 2001, Andrén 2004). Moreover, the great crested newt is vulnerable to the presence of fish (Beebee 1985, Karlström 1995, Joly et al. 2001), and depends on fine-leaved water vegetation for egg laying (Miaud 1995), suggesting that ponds containing this species in general are fish free and provide suitable plant substrates necessary for its reproduction.

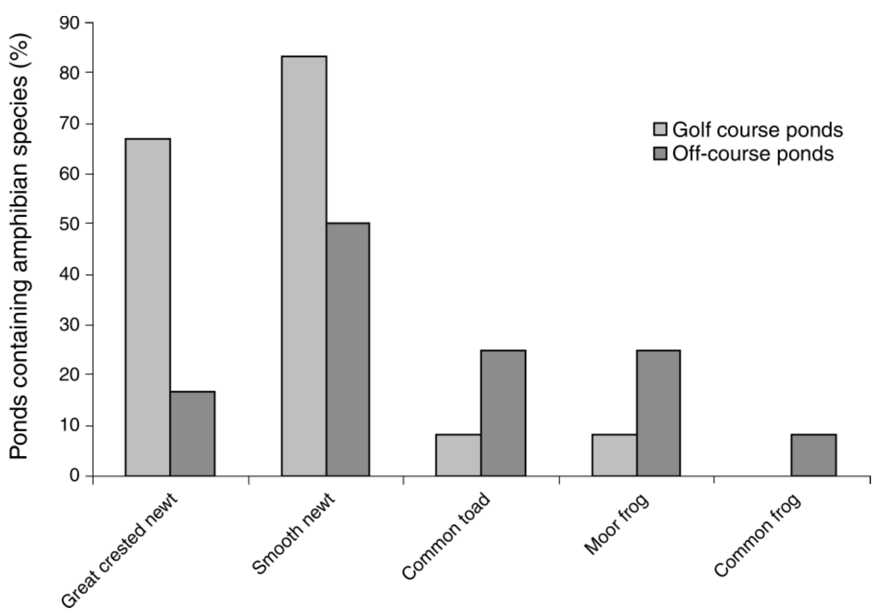

FIG. 3. Amphibians recorded in the case study, by pond location on or off golf courses.

TABLE 1. GIS data for pond distribution in central greater Stockholm.

Location	Land area (ha)	Number of ponds		Aggregated pond area (ha)	
		Total	In this study	Total	In this study
Golf clubs, $n = 13$	571.7	44	12	4.4	0.92
Nature-protected, $n = 16$	4766.2	21	5	3.6	1.60
Miscellaneous, including residential land	45 370.7	102	7	31.0	2.13
Total	50 708.6	167	24	39.0	4.65

While golf course ponds appear to be highly suitable habitats for newts, adult anurans could only be detected in one of the golf course ponds of the survey, containing the common toad and the moor frog. This pond lacked the great crested newt. More generally, no anurans were found in ponds in which the great crested newt occurred. This may be due to the fact that the great crested newt is a known predator of anuran tadpoles, among others, which means that frog survival can be lower in cases of co-existence between anurans and newts (Karlström 1995). This effect has also been described for other species of newts (e.g., Wilbur et al. 1983). In addition to the threatened great crested newt, which thrived in golf course ponds, the large white-faced darter dragonfly also represents an internationally red-listed species that has declined greatly in many parts of Europe and has high conservation priority in the EU Habitats Directive (the Council Directive 92/43/EEC). This species could only be found in three geographically separated golf course ponds. Based on overall results of this study, we provide four major assertions regarding golf courses and wetland fauna.

Chemicals on golf courses

The use of chemical applicants on golf courses in central greater Stockholm does not seem to have a negative effect on the organism groups examined. This finding is quite unexpected, considering that Swedish golf courses are intensively managed with chemical applicants. For example, recommended nitrogen treatment of putting greens is in the range of 150–300 $kg \cdot ha^{-1} \cdot yr^{-1}$, usually distributed 8–15 times, and in the range of 0–150 $kg \cdot ha^{-1} \cdot yr^{-1}$ for fairways (Golfsportens miljöpåverkan 2000). Phosphorus is usually only applied on Swedish golf courses during construction, while potassium is applied regularly and in as large a quantity as nitrogen (Golfsportens miljöpåverkan 2000). Regarding pesticides, fungicides are almost exclusively used on putting greens to treat fungi, predominantly *Monographella nivalis*. Herbicides are used to treat weeds on fairways, mainly white clover (*Trifolium repens*). In comparison, recommended use of nitrogen on intensively managed farmland is 175–225 $kg \cdot ha^{-1} \cdot yr^{-1}$ (Jordbruksverket 1997). These levels have been shown to negatively influence amphibian occurrence in greater Stockholm (Karlström 1995).

TABLE 2. Pond parameters and biodiversity indices for assessed macroinvertebrates (identified to species level).

Ponds	Area (m^2)	Distance to nearest forest patch (m)	Number of species, S	Total individuals (n)	H' (log 10)/log(S)	Shannon index H' (log 10)
G1	500	38	7	231	0.32	0.62
G2	70	17	6	369	0.49	0.89
G3	580	29	10	1191	0.24	0.56
G4	1440	75	14	994	0.50	1.32
G5	2000	72	12	311	0.52	1.28
G6	1060	55	16	4301	0.16	0.44
G7	1330	50	13	453	0.68	1.75
G8	90	10	16	546	0.62	1.72
G9	140	12	15	1457	0.44	1.20
G10	1130	1	10	494	0.63	1.44
G11	800	15	17	1060	0.75	2.13
G12	90	10	8	279	0.73	1.51
OP13	5930	1	15	162	0.74	2.00
OP14	3220	7	16	586	0.62	1.71
OP15	2840	1	10	841	0.36	0.82
OP16	50	30	10	378	0.54	1.24
OP17	550	49	6	119	0.62	1.11
OP18	6240	1	17	519	0.64	1.83
OP19	3910	25	7	633	0.59	1.15
OP20	1750	1	25	586	0.66	2.14
OP21	1210	1	3	11	0.91	0.99
OP22	110	1	3	11	0.55	0.60
OP23	10 350	90	18	972	0.37	1.07
OP24	1090	7	15	437	0.46	1.24

Note: A total of 24 ponds were selected for analysis of fauna: 12 golf course ponds (G) and 12 off-course ponds (OP).

There might be several reasons why fertilizers applied on the golf courses studied do not impact wetland fauna negatively. One reason is that golf course ponds examined were randomly selected in this study, which means that we did not differentiate among golf course ponds adjacent to putting greens (which are more intensively treated with chemicals) and ponds positioned on fairways or out-of-play areas on the golf course. This means that we might have missed golf course ponds that potentially were more contaminated than others. On the other hand, excessive chemicals may eventually reach ponds on a golf course over time through drift outside of the intended areas where they are applied, or they may percolate, or leach, through the soil, as well as be carried to ponds as runoff (States of Jersey 2007).

Another reason is that Swedish golf courses represent habitats with a considerably longer yearly period of plant cover relative to intensively managed arable land (e.g., cropland), which is barren for long intervals, i.e., in the winter and fall. This means that nutrient uptake by plants is considerably greater on a golf course. Such buffering effect of vegetation cover may explain why fertilizers on the golf courses assessed do not reach harmful levels for wetland fauna. It should be recognized, however, that chemical applications on golf courses can promote amphibian tadpole survival. For example, studies from the United States show that insecticides may increase food resources for amphibians through reduced interspecific competition between amphibians and aquatic insects (Semlitsch et al. 2007). Moreover, insecticides can lead to a reduction of insect predators that consume amphibian eggs and larvae (Semlitsch et al. 2007). The use of insecticides on Swedish golf courses is, however, limited and usually only applied to treat frit flies (Chloropidae) on putting greens (Golfsportens miljöpåverkan 2000).

Golf course pond management

A second assertion we make is that golf course pond management benefits groups of wetland fauna. The natural fate of all bodies of standing freshwater is to fill with sediment and vegetation and gradually change to terrestrial habitat (Gee et al. 1997). However, due to aesthetic ideals and in order to fulfil high playing standards, golf course ponds are regularly maintained through the removal of vegetation, preventing natural succession from reaching the stage where water bodies become overgrown and ultimately drained. As confirmed in talks with greens keepers at the golf courses assessed, this practice is routinely conducted on Swedish golf courses. This practice benefits some amphibians (Marsh and Trenham 2001) and macroinvertebrates, e.g., odonates (Schindler et al. 2003). For instance, the two species of newts recorded in this study depend on open water areas for successful mating behavior (Hedlund 1990). Moreover, most Anisoptera species depend on sunny biotopes with a high percentage of exposed macrophytes (Samways and Steytler 1996). In addition to open water areas in ponds, removal of vegetation allows for continuous uptake of phosphorus and nitrogen by fast-growing plants such as cattail (*Typha* spp.) and the common reed (*Phragmites australis*), which were frequent in the ponds of this study. This practice contributes to nutrient retention because plant material is continuously harvested and removed, lowering eutrophication, which is considered to be one of the major impairments of small standing water bodies (Brönmark and Hansson 2002), with associated negative effects on amphibians (Andrén et al. 1988, Berger 1989, Oldham et al. 1997, Camargo et al. 2005). Invertebrate communities are strongly influenced by nitrate levels in ponds (Briers and Biggs 2005), with increased eutrophication leading to a reduction in the number of odonates (Lenz 1991).

While permanent water bodies may favor some groups of wetland fauna, golf courses should ideally also contain seasonal wetlands to optimize their value for fauna more generally due to the absence of predatory fish. For example, Paton and Egan (2003) and Scott et al. (2002) found that golf course ponds with a short hydroperiod tend to have unique amphibian species compared with permanent ponds. Temporary waters on golf courses may also benefit odonates. While most Anisoptera species preferentially breed in lentic, permanent waters (Brooks 1999, Hofmann and Mason 2005), many odonates avoid predators by using habitats that are too ephemeral for the predators to complete their life cycles (Wellborn et al. 1996, Johansson 2000, Johansson and Suhling 2004).

Golf courses and terrestrial habitats

A third assertion that can be made from this study is that golf courses likely also provide suitable terrestrial habitat for wetland fauna. Most pond-breeding amphibians reside in terrestrial habitat patches near breeding ponds for feeding, shelter, and hibernation (Paton and Egan 2003, Rubbo and Kiesecker 2005), and given the philopatric behavior of amphibians, we assume this relationship also holds on the golf courses surveyed. In this study, the recorded maximum distance between any golf course pond and its closest natural forest or shrub patch was between ≤ 1 and 75 m (with a mean distance of 32 m), a range falling well within the known movement ranges of assessed amphibians (i.e., 400–2000 m [Andrén 2004]). That amphibians actually use these terrestrial habitats needs, however, to be confirmed through active search surveys, although such detection is extremely difficult (Semlitsch and Bodie 2003).

Besides amphibians, the golf courses surveyed likely provide suitable terrestrial habitats for a great many of the aquatic invertebrates, e.g., odonates. It is generally known that most of the mature adult life span of odonates is spent at the breeding site (Purse et al. 2003). Zygopterans are generally weak flyers, which tend to occupy the interior of emergent vegetation stands and deposit fertilized eggs among the stalks of wetland vegetation (Foote and Rice Hornung 2005). In general they do not disperse far from their larval habitat, and the majority of mature adults at a pond have generally emerged from the same pond (Bennett and Mill 1995, Hardersen 2000). Adult Anisoptera, in contrast, may disperse considerably longer distances (Conrad et al. 1999).

Urban golf courses and wetland fauna

A fourth, and perhaps the most important assertion, is that golf courses located in urban areas have the potential to provide important habitats for declining groups of wetland fauna (Hodgkison et al. 2007). As revealed in this study, golf courses provide over a quarter of all permanent freshwater ponds that exist in greater Stockholm. This is a considerable resource, given that golf course ponds appear to provide habitats that are as suitable for wetland fauna as ponds in nature-protected areas. As urban wetlands in general tend to have less surrounding forest cover and a greater road density than rural wetlands (Rubbo and Kiesecker 2005), with isolation of amphibian populations (by geographic distance and/or presence of road traffic) as an important factor behind amphibian declines in greater Stockholm (Karlström 1995, Löfvenhaft et al. 2002, Löfvenhaft et al. 2004), golf course ponds are generally embedded within a coherent belt of green cover. Because the golf courses assessed represent large seminatural ecosystems, often containing a whole system of ponds, creeks, and ditches, they represent a vital refuge for local aquatic fauna populations and likely contribute to sustaining larger, regional metapopulations (Hanski 1998, Alford and Richards 1999). Furthermore, one-fifth of all golf courses in greater Stockholm are located adjacent to nature reserves (Colding et al. 2006). Given that exchange of local wetland populations occurs between these land use types, golf courses may provide important buffer habitats near reserves (Colding 2007).

Conclusions

Based on results of this study, we conclude that golf course chemicals on surveyed courses do not seem to impact aquatic fauna negatively; active golf course pond management can benefit some wetland fauna groups; the golf courses assessed likely provide terrestrial habitat for a great deal of wetland fauna; and golf courses provide a substantial amount of wetlands in urban settings. We do not suggest that golf courses in general benefit wetland fauna. However, golf courses with ample wetlands contained on them can significantly contribute to wetland fauna support, particularly in urban settings where green areas are diminishing, and loss of aquatic habitat occurs (Rubbo and Kiesecker 2005). Given that ecological premises are more widely accounted for in golf course design and management, the sport of golf could increasingly become an asset in ecosystem management and biodiversity conservation. For this to be realized, it is essential that ecologists cooperate more closely with urban planners, ecosystem managers, and golf course designers.

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