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ORIGINAL PAPER

Low extinction risk in the flower fly fauna of northeastern North America

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Abstract

Introduction Flower flies (Diptera: Syrphidae; also known as hoverflies) are important pollinators of wild and cultivated plants. Other pollinators such as bees have declined, and many flower flies in Europe and Chile have been documented to be threatened with extinction. The status of other flower fly faunas is currently unknown.

Aims/Methods We assessed the rangewide conservation status of flower flies that occur in Northeastern North America where there is a diverse fauna of 323 native species. Over 150,000 records, drawn from a locality database compiled for a recently published field guide, additional museum records, recent field surveys, and citizen science records, informed the assessments.

Results We found that a minimum of 11 species are at risk of rangewide extinction, 267 have lower extinction risk, and 45 had insufficient data to assess. Our best estimate is that 4.0% of species are at risk, assuming data-insufficient species are at risk at the same rate as data sufficient species. The range for this estimate is 3.4–17.3% at risk, assuming that none or all data-insufficient species are at risk, respectively.

Discussion Factors causing extinction risk in the fauna we studied are poorly known, although habitat destruction likely explains the decline in one species. While at-risk species mostly have saprophagus or brood parasitic larvae, trophic relationships are confounded by phylogeny (the subfamilies Eristalinae and Microdontinae account for most saprophagus or brood parasitic species). The broad geographical ranges of most species likely contributed to the low rate of imperilment.

Implications for insect conservation The small percentage of at-risk flower flies in northeastern North America bodes well for the health of ecosystems there. The results contrast with the situation in Europe, underscoring geographic heterogeneity in flower fly conservation status.

Keywords Conservation status · Diptera · Larval biology · Pollinators · Syrphidae

Introduction

Insect pollinators are well documented for their importance to plant sexual reproduction (Ollerton et al. 2011) and human food production (Klein et al. 2007; Lautenbach et al. 2012). In the past two decades, declines of pollinating insects have been increasingly reported to the extent that pollinator loss is now an important component of the current biodiversity crisis (Potts et al. 2010, 2016; Swengel and Swengel 2015; IPBES 2016, 2019).

In North America, several studies report widespread or localized declines in bee (Cameron et al. 2011), butterfly (Forister et al. 2021), and moth (Young et al. 2017) faunas. However, less attention has been directed at the status of flower flies (Diptera: Syrphidae; also known as hoverflies), which are especially important members of pollinator

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communities in eastern and northern North America (Larson et al. 2001; Skevington et al. 2019; Chisausky et al. 2020) and other biogeographic regions (Ssymank et al. 2008; Rader et al. 2020). An estimated 812 species occur in North America (Miranda et al. 2013). In addition to the pollinating services provided by adults as they forage for nectar and pollen, larvae of many species provide additional ecosystem services. For example, flower fly larvae are documented to be predators of pest aphids and scale insects, plant feeders, bacterial filter feeders (in rotholes or sapruns in trees, as well as playing key roles in sewage lagoon management), and at least one species is a parasitoid (Miranda et al. 2013; Skevington et al. 2019).

Despite their diversity and ecological roles, the conservation status of North American flower flies is somewhat poorly known compared to other regions where the group has been more completely assessed. Globally, assessments for 303 species are available on the IUCN Red List of Threatened Species, with 62.5% threatened (IUCN 2022). All but one of these species occur in Europe, with some extending their ranges to other continents. Other studies on European faunas have identified many species that have declined and are threatened with extinction (Sullivan et al. 2000; Miličić et al. 2017; Powney et al. 2019; Speight 2020; IUCN 2022; Vujić et al. 2022). In a regional, comprehensive assessment, in which only populations occurring in Europe were considered, 37.2% of 890 species were found to be threatened with extirpation on the continent (Vujić et al. 2022). A review of Chilean species found that 33.3% of the 132 species occurring there are threatened with extirpation from the country (Barahona-Segovia et al. 2021). These studies suggest that species elsewhere may also be at risk.

The Canadian fauna was assessed for conservation status in both 2016 and 2022 (CESCC 2016, 2022). These assessments, which covered only the Canadian portion of the ranges of species that occur more widely, found that 55 species (10.7% of the 513 native, regularly occurring species assessed) were at some risk of extirpation from the country, 259 species (50.5%) were not apparently at risk, and for 199 species (38.8%), the risk was unknown due to a lack of sufficient information (CESCC 2022). To our knowledge, no conservation status assessments have been conducted on flower fly faunas in the US or Mexico.

Although targeted, statistically robust, and geographically broad monitoring studies of insect species are rare, especially in North America (Young et al. 2017), growing interest on the part of citizen naturalists and platforms such as iNaturalist (iNaturalist 2022) increasingly allow estimates of extinction risk in some insect groups. Many flower flies are identifiable by photographs, and a recently published regional field guide (Skevington et al. 2019) contributes to a growing number of digitally available observation records to support status assessment. For example, as of 28 February 2022, there were 142,381 species-level flower fly observations from Canada and the United States posted on the iNaturalist platform (iNaturalist 2022). If care is taken to address biases in citizen science data, these observations can be invaluable for evaluation of conservation status and other assessments (Young et al. 2019; Rapacciuolo et al. 2021).

Taking advantage of these newly available resources, we conducted a comprehensive assessment of the conservation status of the flower flies in northeastern North America, a region with a diverse fauna of flower flies. Acknowledging the limited information available for assessments, we grouped species into three categories – at risk, not at risk, and unknown – and examined overall risk and risk status by state or province, taxonomic subfamily, and larval trophic guild. With this information, we hope to inform strategies for conservation actions and to document current gaps in our knowledge.

Methods

Our study area was New York State and the New England states (Connecticut, Rhode Island, Massachusetts, Vermont, New Hampshire, and Maine) in the United States and the Atlantic Canada provinces (Newfoundland and Labrador, New Brunswick, Nova Scotia, and Prince Edward Island) (Fig. 1). Our focal taxa were the native species of flower flies that occur in this area, excluding vagrants (e.g., *Scaeva affinis* and *Allograpta exotica*). In addition, we included 7 taxa listed by Skevington et al. (2019) that have yet to be formally described. For the species occurring in the study area, we assessed the conservation status of their global ranges, including the portions of their ranges outside of the study area.

We obtained flower fly locality data from the specimen database assembled for the Skevington et al. (2019) field guide; records from the Global Biodiversity Information Facility (GBIF.org 2020), iNaturalist (2022), BugGuide (VanDyk 2021), the Atlantic Canada Conservation Data Centre (ACCDC 2022); primary literature; and targeted databasing of 72 species we thought a priori might be at risk, performed in 2019 at the following museums: American Museum of Natural History, Carnegie Museum of Natural History, Cornell University Insect Collection, National Museum of Natural History, New York State Museum, Philadelphia Academy of Natural Sciences, and Yale Peabody Museum; records for Connecticut and Rhode Island from the private collection of Chris Meier and records for Tennessee, Alabama, and Illinois from the California State Collection of Arthropods. In total, we examined over 150,000 records. All locality data were considered, regardless of whether they were from the study area, to enable estimates of global range extent and population trends. We recognize that nearly all of the museum collections consulted are located in the northeastern US, which may have somewhat biased the geographical distribution of the records examined to this region.

The characteristics necessary to identify some species are not typically captured in field photographs, meaning many photographic records were omitted from the study. In general, photographs of species coded by Skevington et al. (2019) as being identifiable by eye were accepted if relevant field characteristics were documented, whereas photographs of species coded as identifiable only through the use of a hand lens or microscope were not used. Examination of photographic evidence available after the Skevington et al. (2019) field guide was performed by JK.

We used the NatureServe conservation status ranking methodology, which categorizes extinction risk in the form of conservation status ranks (Master et al. 2012; Faber-Langendoen et al. 2012). We used the NatureServe rank calculator (NatureServe 2020), a tool that facilitates use of the NatureServe methodology, to derive ranks for each species assessed. The method uses 10 factors in three areas: rarity, threats, and trends; all scored factors are used to derive an output (Table 1). The method outputs a global conservation status rank on a 1–5 scale ("G1" = critically imperiled, "G2" = imperiled, "G3" = vulnerable, "G4" = apparently secure, and "G5" = secure; the "G" indicates a global rank). Species without recent records that are possibly extinct are designated "GH" and those with insufficient information are classified "GU." Uncertainty in factor scores can lead to a "range" rank that spans up to two values (e.g., G1G3 or G2G4). Here, we binned conservation status ranks into three status categories to simplify analysis: at risk (GH, G1, G2, G3, and range ranks where the average rank is included in the range), secure (all other numbered ranks) and unknown (GU). NatureServe status ranks are much more widely used by state, provincial, and federal natural resource management agencies in our study area than other assessment methods such as the IUCN Red List (IUCN 2012; Faber-Langendoen et al. 2012).

Information for the flower flies in our study area was available for scoring only 5 of the 10 factors: range extent, number of occurrences (i.e., areas where a species is present that are somewhat isolated from other such subpopulations, using a minimum separation distance of 10 km between occurrences), environmental specificity, overall threat impacts (rarely used), and long-term population trend. For many species, we could score only a subset of these factors. The NatureServe methodology requires information for at least two factors to derive a status rank (range extent and number of occurrences, or either of these two plus either threat impact, environmental specificity, or one of the trend factors).

Range extent for each species was estimated using a minimum convex polygon (the same as for extent of occurrence calculations in the IUCN Red List) around recent (2000–2021) and historical (pre-2000) records. A range of possible values for the current extent was used where available data showed the recent range extent was much smaller than the historical one, to account for the possibility that insufficient search effort caused the apparent decline. The NatureServe system supports inclusion of uncertainty by allowing entry of a range of values for a factor, although range values cannot span all possible factor values. We were able to estimate range extent for all but 14 species.

For number of occurrences, a plausible range of values was based on the number of confirmed recent occurrences. For example, *Platycheirus thompsoni* occurs from Minnesota and northwestern Ontario in the West to Pennsylvania and the Maritime provinces in the East. It is a small, difficult-to-identify species typically found in peatlands, and generally cannot be identified from field photographs. We found nine recently documented occurrence records from across much of the known range. Based on the amount of available habitat across the range that has not been surveyed adequately, we estimated that at least 20 occurrences truly

Table 1 Factors and weightings used to calculate conservation status ranks. Factors in italics are used only if information for other factors in category is unknown. The weighted average of the rarity and threats sub-scores generate an initial score, which is then modified up or down using the trends sub-score. The final score is compared to thresholds to assign an overall status rank. See Faber-Langendoen et al. (2012) for more details

Factor Category	Factor Category Weight	Factor	Factor Weight
Rarity	0.7	Range extent	1.0
		Area of occupancy	2.0
		Population size	2.0
		Number of occurrences	1.0
		Number of occurrences with good viability	2.0
		Environmental specificity	1.0
Threats	0.3	Threat impact	1.0
		Intrinsic vulnerability	1.0
Trends		Short-term trend	2.0
		Long-term trend	1.0

exist, and that possibly more than 300 exist (thus selecting the rating values 21–80, 81–300, and over 300).

Environmental specificity considers a species' degree of specialization and the availability of key resources in the environment. In flower flies, larvae are generally far more specialized than adults, so the larval life history was used to derive the environmental specificity score. Information about larval life history was derived from published literature, primarily Skevington et al. (2019).

Current threats to flower fly species are not well understood and therefore overall threat impact was rarely scored.

Long-term population trend was estimated by comparing the range extents and number of documented occurrences of the recent and historical periods for each species. We assumed that changes in range extent or number of occurrences reflected changes in population numbers. Because there have been no systematic surveys done to determine population trends of flower fly species in our study area, precise trend values were not assigned. We addressed uncertainty by determining plausible values for factors (IUCN Standards and Petitions Committee 2022). For example, if the range extent declined substantially between the two eras (e.g., > 50%) then a range of values was used (e.g., decline of 0–70%). This accounts for the possibilities that the change was due to a true population decline or an artifact from insufficient recent sampling. Whether a species can be identified from photographs posted to citizen science platforms was considered - there are far more recent records available for easily identified species, so we assumed that sampling was more complete for them.

To provide a comparison with the IUCN Red List categories, we followed the Red List guidelines (IUCN Standards and Petitions Committee 2022) to suggest a range of plausible Red List categories for each species determined to be at risk. Data were sufficient to assess criterion B2ab (small extent of occurrence, few locations, and continued decline) only. We applied the maximum and minimum range extent values used to calculate NatureServe status ranks to the extent of occurrence values. Approaches to estimating other Red List parameters matched those used for the NatureServe status ranks.

Analyses. We first calculated the percentage of species falling into each of the status categories and then tallied the results by state or province. To investigate the influence of larval biology on conservation status, we summarized conservation status categories by species' larval trophic guild (brood parasite, predator, phytophage, saprophage, or unknown) as described in Skevington et al. (2019). Similarly, we investigated the effect of phylogenetic identity by summarizing conservation status category by subfamily, following the widely accepted taxonomy used by Skevington et al. (2019) and, separately, the subfamilies proposed by Moran et al. (2022). Too few species were categorized as at risk for us to be able to perform statistical analyses.

Results

The focal area's native, resident flower fly fauna consists of 323 species (about two-fifths of the entire US and Canadian fauna). The full list of species with taxonomic group (subfamily and operational taxonomic unit affiliation), larval trophic guild, conservation status, and subnational distribution information is provided in the Supplementary Information. The number of species documented per state or province averages 158 species, ranging from 51 species (Rhode Island) to 232 species (New York) (Fig. 1).

We found 11 species to be at risk, 267 secure, and 45 with too little information to determine status. Estimates of the percentage of species at risk ranged from 3.4% (assuming all unknown species to be secure) to 17.3% (assuming all unknown species to be at risk), with a best estimate of 4.0% (assuming unknown species to be at risk at the same ratio as the known species). We provide details about the status of the at-risk species in Table 2. The number of at-risk species documented for each state or province ranged from 0 (Newfoundland and Labrador, Prince Edward Island, and New Brunswick) to 6 (New York) (Fig. 1).

The global ranges of two at-risk species are largely limited to the study area. *Mixogaster johnsoni* is associated with sandy habitat and is known from few records in the southern coastal portion of the study area, south to New Jersey, just south of New York. *Chalcosyrphus aristatus* is known from one state in our study area (New Hampshire) and one state adjacent to the study area, Pennsylvania. Further survey work may prove this species to be more widespread than currently available records suggest. The remaining at-risk species have been recorded more broadly from eastern North America or, in the case of *Chalcosyrphus depressus*, North America.

At-risk flower flies have larvae that are saprophagous, brood parasites, or predators (Fig. 2). None of the species with phytophagous larvae was assessed to be at risk, although 6 of these species had unknown conservation statuses (Fig. 2).

We found at-risk species among the members of the Eristalinae (7 species), Microdontinae (3 species), and Syrphinae (1 species) subfamilies, regardless of the taxonomic framework followed (Fig. 3).

Table 2 At-risk flower flies occurring in northeastern North America. Plausible IUCN Red List categories are provided for comparison ("CR" = Critically Endangered, "PE" = Possibly Extinct, "VU" = Vulnerable, NT = Near Threatened, "LC" = Least Concerned). The criteria met or nearly met for all categories except LC was B1ab(iii) (small extent of occurrence, few locations, and continuing decline in inferred extent and/or quality of habitat; IUCN 2012)

Species	Status Rank	Plausible Range of Red List Categories	Comment
Anasimyia distincta	G2G3	NT-LC	Limited historical distribution from Virginia to southern Ontario. Records in the 2020s from Maryland and Virginia are the first since 1983. *Anasimyia* are habitat restricted. Only males are identifiable.
Chalcosyrphus aristatus	G1G3	CR-NT	Known from only 9 records and as few as four occurrences in Pennsylvania and New Hampshire; most recent in 1983. Males are distinctive, although specimens are required to confirm identification.
Chalcosyrphus depressus	GH	CR(PE)	Historically widespread, with records from eight sites in British Columbia, Ontario, Quebec, Idaho, New Hampshire, and Wyoming. Not collected since 1951. Major western North American museums have not been databased, so it is possible more recent records exist. Citizen science records are possible (field photos of males should be identifiable), but none exist.
Chrysogaster inflatifrons	G2G4	VU-LC	Recorded historically from the lower Mississippi River to Maine, with an isolated record in Wyoming. This species likely declined substantially in the 20th century. Skevington (2021) documents 112 records, with the most recent in 1986 (Mississippi); all other records predate 1970. Additional records are from Connecticut (1978), Illinois (1999), and Alabama (2002; although this specimen may represent an undescribed species; M. Hauser, pers. comm. to JK, 2022).
Hadromyia aepalius	G2G4	VU-NT	Historically recorded from Georgia to Ontario. The species is rarely collected, with only 2 post 2000 records (Pennsylvania in 2018 and Tennessee in 2020) compared to 8 historical ones.
Leucozona xylotoides	G2G4	NT	A distinctive but rarely documented species known from Virginia to southern Quebec and Maine. A lack of recent records from most of range (e.g., historical records from Virginia and Pennsylvania, but recent records only from Connecticut (2006, 2008, 2009), Maine (2020), and southern Quebec (2021) suggests a possible decline.
Microdon fuscipennis	GH	CR(PE)	This species is known historically from New Mexico to Florida and north to Quebec. The most recent record is from 1997 (Louisiana). The species is distinct enough that citizen scientist photographs should be identifiable, yet none exist in either iNaturalist or BugGuide. Although *Microdon* spp. can be overlooked because they do not visit flowers, there are nearly 300 records from Canada and United States on iNaturalist for congeners, suggesting that the lack of *M. fuscipennis* records indicates a true population decline.
Mixogaster breviventris	G3G4	NT-LC	A widespread eastern North American species. It is uncommon - there are only about 11 occurrences known. The only post 2000 records are from the western part of the range (Oklahoma in 2021, Iowa in 2021, Nebraska in 2018, Arkansas in 2019), suggesting some decline has occurred.
Mixogaster johnsoni	G1G3	CR-VU	An Atlantic Coastal Plain species associated with sandy habitat (Maier 2011) and known from New Jersey to Massachusetts. Few records exist, with only one since 2000, in 2009 (Rhode Island).
Sericomyia slossonae	G2G3	NT	A rarely collected species with a small distribution (east of Lake Superior to Maine). Since 2000, it has been documented only in Ontario (2001 and 2016) but presumably persists elsewhere such as northern Maine where it was collected in the 1990s and few recent surveys have taken place.
Tropidia calcarata	G3G4	NT-LC	Known from 11 occurrences ranging from South Carolina to Indiana and eastern Quebec where it is found in wetlands with yellow pond-lilies (*Nuphar advena*). Only two occurrences have been documented since 2000 (Pennsylvania in 2007 and 2010 [same site] and New Jersey in 2021), suggesting a decline.

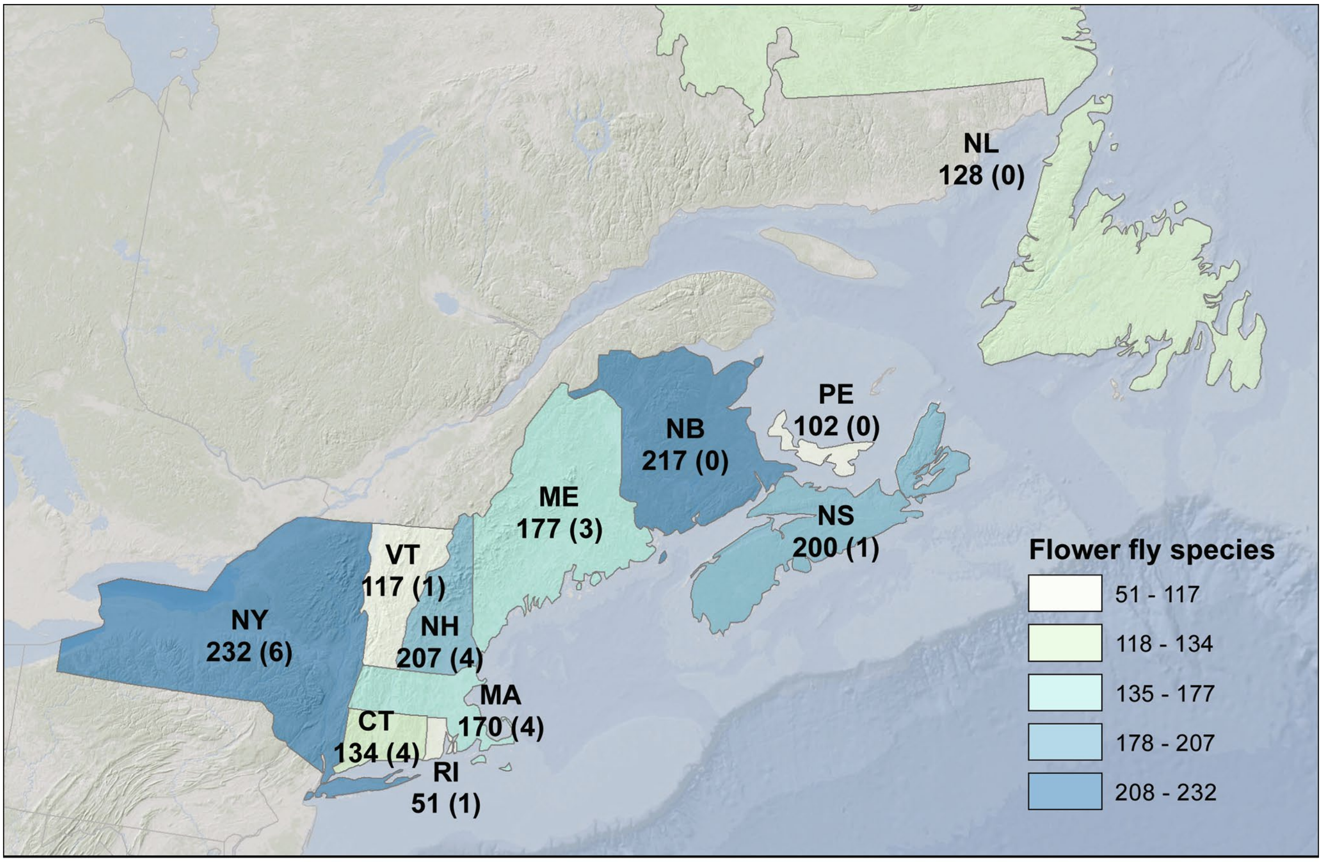
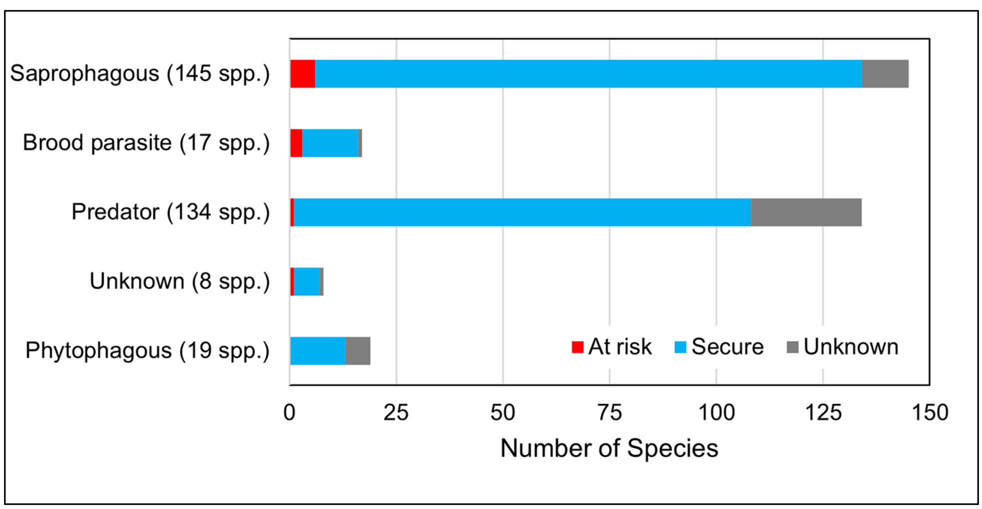


Fig. 1 Species richness and numbers of at-risk species (in parentheses) for flower flies in states and provinces in northeastern North America. Abbreviations: CT, Connecticut; MA, Massachusetts; ME, Maine; NB, New Brunswick; NH, New Hampshire; NL, Newfoundland and Labrador; NS, Nova Scotia; NY, New York; PE, Prince Edward Island; RI, Rhode Island; VT, Vermont

Fig. 2 Conservation status of northeastern North American flower flies by larval trophic guild. Numbers of species in each guild listed in parentheses



Discussion

To our knowledge, this is one of the first studies to comprehensively assess the rangewide conservation status of a large regional flower fly fauna outside of Europe. Primarily relying on museum, field, and citizen science data, we were able to assign 86.1% of the 323 species in our study area into at-risk or secure groupings. Moreover, we compiled modern species lists for the 7 US states and 4 Canadian provinces that make up our study area. Remarkably, our best estimate is that only 4.0% of species appear to at risk of extinction,

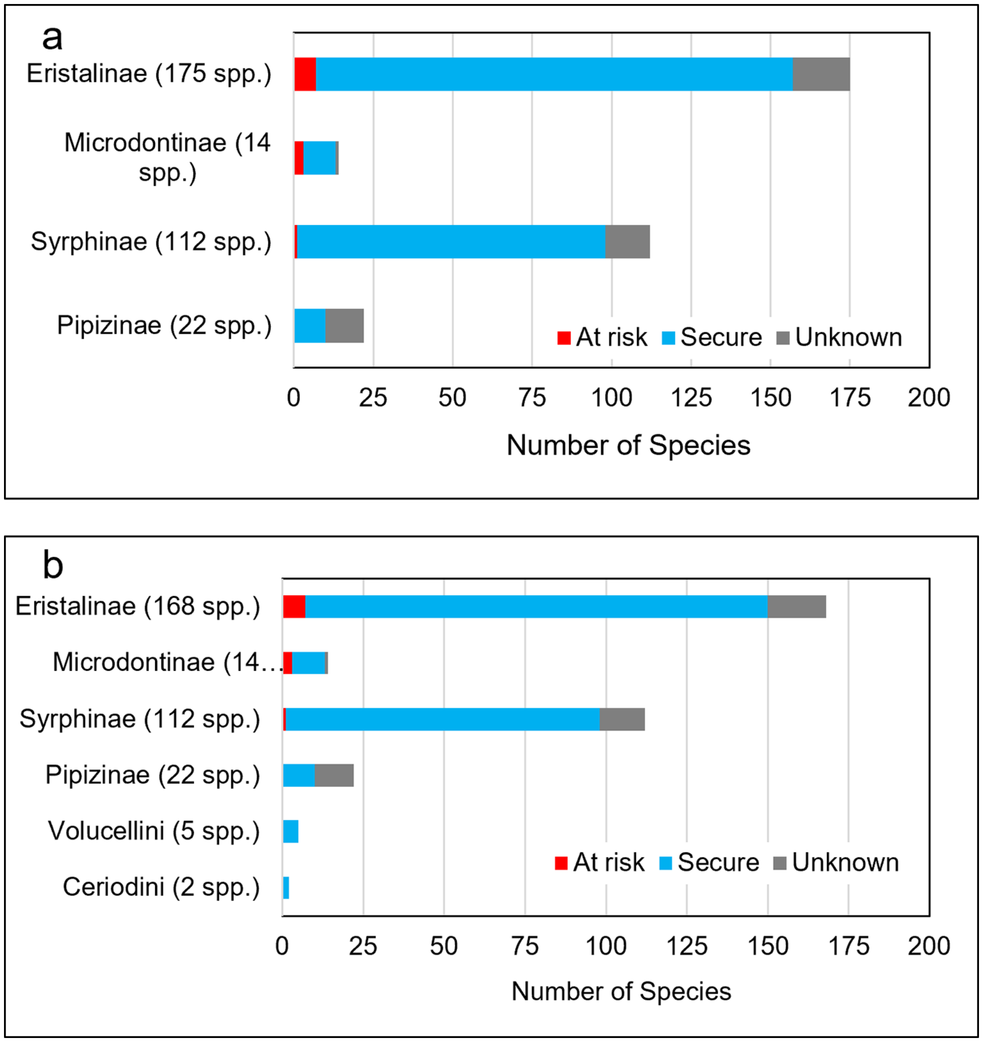


Fig. 3 Conservation status of northeastern North American flower flies by phylogenetic group. a, traditional subfamilies (i.e., those used by Skevington et al. 2019); b, operational taxonomic units defined by Moran et al. (2022). Numbers of species in each group listed in parentheses

contrasting with other summaries of the conservation status of pollinators (Potts et al. 2016; IPBES 2016).

Although digitized information useful for assessing the conservation status of flower flies in our study area has increased in availability recently, our work should be considered preliminary. Collections and citizen science data that were not collected in a systemic manner aimed at documenting distribution or population trends create challenges to data interpretation (Young et al. 2017, 2019; Rapacciuolo et al. 2021). For this study, specimen data were the basis for determining historical range extent and abundance whereas the majority of records that informed current estimates of these measures were from citizen scientists. Some issues, such as flower-visiting species being more commonly encountered than those that do not visit flowers, may be similar for both specimen collectors and citizen scientists. However, citizen scientists might be more likely to make observations near urban areas or overlook rare species (Kosmala et al. 2016; Young et al. 2019). Also, the increasing number of observations available today could make species appear more abundant than historically. We did not use area of occupancy as ranking factor due in part to its high sensitivity to sampling effort (IUCN Standards and Petitions Committee 2022). Overall, by using ranges of plausible measures for each conservation status ranking factor and lumping assessment outputs into just three broad status categories, we attempted to minimize possible errors in estimating extinction risk caused by these potential biases.

Assessments for most species in this study were derived from few status ranking factors and many factors were scored with a range of values reflecting uncertainty. We hope that our results inspire broader data collection, data sharing, and research that can lead to refinements in the future. For example, current threats to flower flies are not well understood (see below) but if future research identifies widespread and pervasive threats, additional species could shift to the at-risk group.

The European flower fly fauna has received a recent regional comprehensive assessment (i.e., consideration of the populations occurring within European boundaries)

(Vujić et al. 2022). That study showed that over a third of species are threatened (i.e., falling into the critically endangered, endangered, or vulnerable categories, which are roughly equivalent to the NatureServe G1-G3 categories; Regan et al. 2005; IUCN 2022), much greater than in our study. With a third of species threatened, the Chilean fauna is also threatened at much greater rates than what we found for northeastern North America (Barahona-Segovia et al. 2021). Concern about declining populations in Europe has led to efforts to create artificial breeding sites and translocating individuals (Rotheray 2010). Northeastern North American forests are not as heavily managed as many forests in Europe, especially in Britain where rotting logs and other breeding sites can be rare (Rotheray 2010). Also, our study area lacks the large mountain ranges of Europe or Chile (where many species have restricted ranges), resulting in most species being widespread and therefore less likely to be categorized as at risk. Indeed, 91.9% of the 309 species in our study with sufficient information had range extents of at least 200,000 km^2.

Our results also contrast with the Canadian study that found 10.7% of flower fly species at risk of national extirpation (CESCC 2022), with a best estimate of 17.5% of the fauna at risk (using the method we used that assumes species with insufficient information are at risk at the same rate as species with sufficient information). Most of the at-risk species in the Canadian study occur in British Columbia, where Canada represents a small portion of the ranges of species that occur more broadly in the western US or where ranges of some species are small in mountainous regions (CESCC 2022). Given the breadth of conservation status information compiled in this study, future Canadian national assessments could benefit from our results for the 262 species that occur in Canada.

Better understanding of the threats facing the flower flies in northeastern North America requires further study. Loss of key habitats such as forests and wetlands, together with invasive species likely pose important threats, particularly in the heavily developed southern portion of the study area. Existing forests have also become degraded after many native trees that once dominated forests such as American beech (*Fagus grandifolia*), American elm (*Ulmus americana*), American chestnut (*Castanea dentata*), eastern hemlock (*Tsuga canadensis*), and ash (*Fraxinus* spp.) have succumbed to introduced pests and diseases (Burnham 1988; Cale et al. 2017; Marks 2017; Valenta et al. 2017; Ellison et al. 2018). Another possibly underappreciated threat is the loss of natural habitat on hilltops, important mating sites for many insects (Alcock and Dodson 2008), including rare flower flies in northeastern North America (Skevington et al. 2019). Destruction of natural vegetation at summits, such as for the construction of communication towers, disrupts "hilltopping" mating behavior (Sands 2018).

The introduced harlequin ladybird (*Harmonia axyridis*), which is common and widespread in the southern part of the study area, is known to prey upon the larva of *Episyrphus balteatus*, a common European flower fly (Ingels and De Clercq 2011), and it may have indirect competitive interactions with aphidophagous flower fly larvae, as it does with other ladybird species (Roy et al. 2016). The impact of harlequin ladybird and similar invasives on the northeastern flower fly fauna is worthy of research. In addition, *Eristalis brousii*, categorized as secure, has declined due to a putative threat. This species' disappearance from the eastern United States in the early 1900s was coincident with the introduction of its European sister species, *E. arbustorum*. Hybridization with that species may have led to its demise there and could be an ongoing threat in areas where *E. brousii* still occurs (Skevington et al. 2019).

Currently no flower fly has legal protection in either the US or Canada. *Eristalis brousii* has been identified as a candidate for detailed status assessment in Canada, which could lead to it being listed under Canada's Species at Risk Act (COSEWIC 2022). We suggest closer review of all 11 species identified here as at risk for consideration for legal protection at either federal or state/provincial levels.

The 45 species of unknown status are all hard to document because most are difficult to identify (e.g., require examination under stereo microscope), are inconspicuous in nature (e.g., ≤ 10 mm long, drab coloration), or a combination of the two (*Parhelophilus brooksi* is an exception). They all have few or no recently documented records, and, with the exception noted above, because they are cryptic, the lack of record documentation could be the result of insufficient search or identification effort. The Pipizinae has the highest proportion of unknown statuses. Nearly all species in this subfamily are small (< 10 mm long), inconspicuous (most are all black), and difficult to identify. As a result, they are often overlooked in the field and cannot be effectively surveyed by citizen scientists. Given the challenges to monitoring these species, assessing their conservation status will continue to be a challenge until genetic identification techniques such as DNA barcoding become more widespread.

The marked differences in state and provincial species richness across the study area were likely influenced by survey effort. Because the intensity of flower fly collecting varies among these jurisdictions, there is greatest disparity in the richness of species that can be identified only by examining caught specimens in the lab. The Maritime provinces (New Brunswick, Nova Scotia, and Prince Edward Island) have been subject to relatively high levels of sampling, both historically and recently. Cape Breton Highlands National Park in Nova Scotia and Kouchibouguac National Park were subject to intensive insect surveys in the 1970s and 1980s, with flower flies being a focal taxon both times (Lafontaine et al. 1987). Of the 3,636 specimen records collected in 2000 or later from the focal area in the Skevington (2021) database, 99.1% (3,605) are from the Maritimes (most collected by JK). Also, New York recently completed a five-year pollinator survey that included flower flies (White et al. 2022; some results were not available in time to inform this study). As a result, the species totals for those four jurisdictions are likely more realistic relative to other jurisdictions.

With few at-risk species, we were unable to unequivocally identify phylogenetic or trophic signatures of imperilment status. Although at-risk species appeared clumped both taxonomically (to Eristalinae and Microdontinae) and trophically (to saprophagy and brood parasitism), we note that phylogeny and life history are not independent. All Microdontinae species assessed are brood parasites, and they comprise 14 of the 17 species in this trophic category. We know little about microdontine host choice, but the subfamily appears to be more host-specific than other predators and thus have localized populations that are more prone to decline (Howard et al. 1990a, b; Schönrogge et al. 2002, 2006; Gardner et al. 2007; Reemer 2013). A similar situation exists with another guild in Europe. Many of the threatened flower flies in Europe are restricted to the Mediterranean region, where they are host-specific, habitat specialists (for example: many of the *Merodon* and *Eumerus* species that feed on bulbs) (Milankov et al. 2007; Francuski et al. 2010; Grković 2021; Janković and Radenković 2021).

Our assessments of northeastern North America are available online where policy makers and others can find them (https://explorer.natureserve.org). Specific recommendations include targeted searches for the at-risk and species with too little information for assessment and broadscale surveys designed to determine species-level population trends. Such programs can be organized by state and provincial non-game wildlife authorities with participation by both citizen scientists (capable of identifying many species using a field guide) and professional entomologists (capable of identifying all species, including those that require capture and examination in the lab) as was done in the recently completed New York survey (White et al. 2022). This information can inform revised conservation status assessments and, in the US, inclusion as species of greatest conservation need in state wildlife action plans (AFWA 2010). These surveys and corollary state-level assessments may reveal species at risk of local extirpation that are secure globally (White et al. 2022). Where species are demonstrably highly threatened, federal wildlife agencies should consider listing under the US Endangered Species Act or Canadian Species at Risk Act.

The potentially low level of imperilment we found for northeastern North American flower flies is a rare bit of good news for conservation in an area characterized by expanding urbanization, few remaining old growth forests, and the decline of keystone tree species. Although populations of secure species may be declining in parts of their ranges and some of the 45 unknown species are likely at risk, overall, the syrphid fauna in the study area appears to be remarkably intact. Population monitoring of these important pollinators must be expanded, but for the time being, a broadscale decline in the diversity of syrphid pollinators appears not to be a top threat facing terrestrial ecosystems in northeastern North America.

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Declarations

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