Aquatic and terrestrial invasives species in the Lake Simcoe watershed: presence, distribution, and vulnerability to future invasions.

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Introduction

Global climate change and the introduction of invasive, non-native species into ecosystems represent two of the most significant threats that are influencing the biodiversity of native ecosystems (Mainka and Howard 2010). A species is described as invasive when, after introduction, its presence causes damage to local biodiversity, human development or human health (Mainka and Howard 2010, Millenium Ecosystem Assessment 2003). Invasive species also threaten human-managed ecosystems (i.e. agriculture, forestry, aquaculture) where estimates of annual economic losses in the U.S. are over 100 billion dollars (Pimentel et al. 2000) and in Canada they are estimated to be over 5 billion dollars (Dawson 2002). Butchart et al. (2010) recently released a synthesis of global biodiversity data which investigated the status of the 2002 Convention of Biodiversity's goal to slow the loss of global biodiversity by 2010. They analyzed data covering species' population trends, extinction risks, state of critical habitat, and community composition and found global biodiversity continued to decrease. A significant contributor to this phenomenon is the increased number of non-native species that are outcompeting local species and altering the structure, function, and disturbance regime of ecosystems (Butchart et al. 2010; Dukes et al. 2009). Compounding the problem further, the trend in new invasions continues to increase in the Great Lakes region (Riccardi 2001) suggesting that there are likely future threats waiting to colonize the Lake Simcoe watershed.

The management of invasive species is a very reactive process where success is limited if agencies are forced to deal with already established populations. Limitations that have been identified include effective management tools, adequate funds, and a general lack of understanding of the biology of the introduced organism (Larson et al. 2011). Thus a key objective of most management programs is to prevent new introductions and identify any potential vulnerabilities which may exist in the management area (Larson et al. 2011). This involves predicting the factors that are responsible for the spread and establishment of invasive species. Even though climate is likely the most important factor that regulates ecosystem properties at larger spatial scales (Chapin et al. 2002), when understanding the immigration and emigration of species at

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smaller spatial scales the influence of climate must be considered in the context of other factors. Lonsdale (1999) states that the degree to which an area is invaded is a function of the properties of ecosystem being invaded (i.e. degree of disturbance and resistance to invasion), the properties of the native species within that ecosystem (i.e. competitive abilities), the properties of the invading species (i.e. suitability of species to invade), and the propagule pressure of the invading species (i.e. number of propagules delivered to the site).

For invasive species that are already present in the region, managers are asked to apply risk assessment models that ultimately help them determine the extent to which management interventions will be applied to contain or eradicate the new invader (see An Invasive Alien Species Strategy for Canada, 2004).

The *Lake Simcoe and its Watershed* report to the Minister of the Environment (LSSAC 2008) identified 12 aquatic invasive species currently present in the region, and 5 or more terrestrial invasive species. However, limited monitoring and reporting, especially of terrestrial species, makes the quantification of the different species difficult. Species reported in the watershed are presented in Table 1. An explicit goal stated in the *Lake Simcoe and its Watershed* report is to ensure that there are no new introductions into the watershed (LSSAC 2008).

Methods

To assess the vulnerability of the Lake Simcoe watershed to future invasive species introductions and the potential influences of climate change on its vulnerability, the following performance indicators are proposed:

1. Presence/abundance of invasive species
2. Health of native biological communities

Due to the absence of quantitative and spatial data sets for invasives in the Lake Simcoe watershed, we are going to assess these indicators against five case studies of invasive species. Dog-strangling vine, Eurasian milfoil, and Emerald ash borer are invasive species already established in the watershed (or nearby, in the case of Emerald ash borer) and they represent plants, aquatics, and forest pests, respectively. Kudzu vine and 1000 cankers disease (affecting Black walnut) are two potential invaders to the watershed. Each of these five species presents a unique perspective on the influence of climate change on their biology and also their interactions with the other disturbance factors present in the watershed.

Table 1: Invasive species present in the Lake Simcoe watershed and their date of introduction, if known.

Aquatic Invasive Species[1]	Terrestrial Invasive Species[2,3]
Common carp (*Cyprinus carpo*) – 1896	Purple loosestrife (*Lythrum salicaria*) – early 1800's
Rainbow smelt (*Osmerus mordax*) – 1962	Himalayan balsam (*Impatiens glandulifera*) - 2009
Eurasian watermilfoil (*Myriophillum spicatum*) – 1984	Dog-strangling vine (Cyananchum species) – mid 1800's
Curly-leaf pondweed (*Potamogeton crispus*) – 1961-1984	Giant hogweed (*Heracleum mantegazzianum*) – 2007
Black crappie (*Pomoxis nigromaculatus*) – 1987	Buckthorn (*Rhamnus cathartica, Rhamnus Frangula*) – late 1700's, widespread by early 1900's
Zebra mussel (*Dreissena polymorpha*) – early 1990's	Japanese knotweed (*Polygonum cuspidatum*) – late 1800's
Spiny water flea (*Bythotrephes longimanus*) – 1993	Garlic Mustard (*Alliara petiolata*) – early settlers
Bluegill (*Lepomis macrochirus*) – 2000	Common reed (*Phragmites australis*) – Atlantic coast in 1800's, westward 1900's
Quagga mussel (*Dreissena bugensis*) – 2004	Emerald ash borer (*Agrilus planipennis*) – 2002 (Ontario)
Rusty crayfish (*Oronectes rusticus*) – 2004	Invasive earthworm species
Round goby (*Neogobius melanostomus*) – 2006	
Echinogammarus ischnus – 2005	

[1] Source for species list and introduction dates: Lake Simcoe and its Watershed: Report to The Minister of the Environment, 2008.
[2] Lake Simcoe and its Watershed: Report to The Minister of the Environment, 2008; Lake Simcoe Protection Plan, 2009; Canadian Food Inspection Agency www.inspection.gc.ca accessed November 2010.
[3] Introduction dates are for North America, unless otherwise indicated. Source: Invading Species Awareness Program and Invasives Tracking System (Ontario Ministry of Natural Resources and Ontario Federation of Anglers and Hunters) www.invadingspecies.com , http://www.comap.ca/ITS/

1. Presence/abundance of invasive species

The presence of an invasive species is an important and fundamental indicator of watershed vulnerability. Despite recent initiatives for preventing invasive species introductions in Ontario, management practices have not been successful at preventing or managing introductions (Hogsden et al. 2007; Claudi *et al.* 2002). The best way to control invasive species is to prevent their arrival. Thus, their occurrence both in the watershed and in the surrounding regions should be carefully monitored to ensure early detection and prevention of threats is possible. Prevention is often accomplished through regional programming that predicts the most likely pathways of introduction. Once identified, monitoring programs are established that target these specific pathways (see A Canadian Action Plan to Address the Threat of Aquatic Invasive Species, 2004).

Once a species becomes established, tracking changes to biomass/density and population expansion becomes very important, as this information feeds into the risk assessment process that is utilized for making management decisions (Allen et al. 2006; Larson et al. 2011). Monitoring of populations will also provide important information as to the biology of the species in its new environment, which will help agencies in formulating a sustainable management plan. Many times there are new morphological and physiological adaptations that arise within the community in its new environment that limit the success of control measures that were successful in the native range (i.e. herbicide tolerance in invasive plants (Richardson 2008))

Climate in general dictates both natural dispersal of species and subsequent success of establishment of reproductive populations. Many terrestrial species use wind or rain to carry seed or cuttings from the host plant to new locations. Aquatic species rely on high water events, such as the spring freshet and storm events, to provide temporary links between environments. Once at this new destination, these seeds or individuals must find conditions suitable for growth and reproduction in order to establish: temperature, moisture, light. All of these factors can be affected by climate change. Alterations to wind, precipitation, and temperature as indicated in the climate change modelling could cause both increased dispersal rates and a change in the species able to establish.

2. Health of native biological communities

The role of disturbance in maintaining the function of natural ecosystems is well documented (i.e. role of fire in a tall grass prairie, water level fluctuations in coastal wetlands). However, it has also been recognized that disturbance, most often anthropogenic in nature, can also decrease the health of ecosystems. This is often recognized by the appearance of such symptoms as high rates of species mortality, excessive productivity, regressive succession, a greater proportion of invasive species, and loss of ecosystem services (Rapport et al. 1998). An underlying assumption of the ecosystem health metaphor is that managers have a sound understanding of the normal rate processes for ecosystems in their region.

There is active debate as to whether invasive species are the direct cause of losses in ecosystem health or whether they are an indirect consequence of external disturbance factors weakening the resilience of local ecosystems (Didham et al. 2005; Gurevitch and Padilla 2004; Hulme 2006). Yet, there is agreement in the literature that where disturbance in ecosystems is high and ecosystem health is compromised, there is a strong correlation with the presence/dominance of invasive species (Roley and Newman 2008; Ramcharan et al. 1992). Changes in climate are an example of such a disturbance that could result in the movement of invasive species into new ecosystems. Parmesan and Yohe (2003) assessed the changes in habitat range for 1700 species and found that there was an average shift poleward of 6.1 km/decade and that spring phenological events were advancing at a rate of 2.3 days/decade. Climate change also creates challenges for the ability of native species to adapt to a new suite of abiotic conditions and thus creates opportunities for invasive species to occupy available niches (Mainka and Howard 2010).

In addition to climate, studies have demonstrated that changes to regional atmospheric chemistry (CO_2, O_3, NO_x, SO_x), physical disturbance to habitat, and landscape fragmentation can all result in changes to the structure and function of native communities and also increase their susceptibility to invasion by non-native species (Allen et al. 2006, Dukes and Mooney 1999; Gavier-Pizarro et al. 2010, Wedin and Tilman 1996).

The Lake Simcoe watershed is renowned for both its summer and winter cottaging, recreation, and fishing, and as such faces high vulnerability for human mediated introductions. As well, it's proximity to the Greater Toronto Region places it at risk for poor air quality during the summer growing season when prevailing winds carry this urban air into the adjacent rural lands (Sager et al. 2005)

Results & Discussion

Dog-strangling vine (DSV)

Dog strangling vine and swallow-wort are common names given to members of the Genus *Cyananchum*, part of the milkweed family (*Asclepiadaceae*) (for an extensive review see DiTommaso *et al* 2005b). All species of DSV share the same general characteristics: herbaceous perennial twining vines, opposite leaf pairs that are ovate or elliptical and tapered at the distal ends, up to 2 meters in height, relying on support from other vegetation or structures. They produce feathery seeds in late summer, which are mainly wind dispersed (DiTomasso *et al* 2005a, OFAH 2010). DSV species are also able to reproduce vegetatively.

There are three species of DSV/Swallow-Wort documented in North America: *C. rossicum* is native to the eastern Ukraine and southwestern Russia; *C. nigrum* (scientific synonym *C. louiseae*) is native to the Mediterranean regions of France, Spain and Portugal (DiTomasso *et al* 2005a,b); and *C. vincetoxicum* (scientific synonym *V. hirundinaria*) native and widespread throughout Europe (Tewksbury *et al* 2002, DiTomasso *et al* 2005b). All of these species were first introduced to North America as cultivated garden plants. Documented occurrences of *C. rossicum* and *C. nigrum* have centered on the northeastern US and eastern Canada, particularly Ontario and Quebec (Sheeley and Raynal 1996, DiTomasso *et al* 2005b). *C. rossicum* and *C. nigrum* are considered to be naturalized in Ontario; whereas, *C. vincetoxicum* has only been confirmed on rare occasions in Ontario related to what is referred to as 'garden escapes' (DiTommaso *et al* 2005b). Both *C. rossicum* and *C. nigrum* can grow in most available habitats in Ontario, excluding wetlands and areas of prolonged flooding, and although it thrives in disturbed areas and open habitats it also establishes well in old fields and forest understory (Lawlor *et al* 2002). *C. rossicum* has been described as an alien vine of major concern in the Lower Great Lakes Basin (DiTommaso et al., 2005b).

Both *C. rossicum* and *C. nigrum* can grow in most available habitats in Ontario, excluding only wetland areas (Lawlor et al., 2002). DSV prefers calcareous soils but can grow in a variety of other soil types. Although the plant thrives best in open habitats, it is now actively invasive in old fields and forest understory. Both species also have the potential to dominate the herbaceous plant forms, however plant densities are generally considerably lower in shady habitats, such as under forest canopies relative to open sunny locations (DiTommaso, et al., 2005).

DSV is well adapted to invasion on both anthropogenic and naturally disturbed sites including quarries, roadside embankments, rail lines, fallow or pasture agricultural lands, as well as naturally eroded stream banks, ravines, and slumps (DiTomasso, et al., 2005b). The vine is able to expand into nearby, less disturbed habitats once it becomes established. The vines form large mono-specific populations in upland habitats and easily adapt to a wide range of moisture and light conditions. Rivers and streams that experience spring flood scouring or areas subject to hydrologic extremes are vulnerable to invasion. (Lawlor,2002). The alvar communities of the eastern Lake Ontario region that occur in the Lake Simcoe watershed are of particular concern as the limestone soils and open nature of the habitat make this rare ecosystem extremely vulnerable.

No known studies have investigated or predicted the impact of climate change on DSV. However, inferences can be made using more general published information regarding changes in various environmental factors on invasive plant growth and viability.

Some studies have identified a clear positive impact of elevated CO_2 and plant resource availability and growth rates (Ziska, 2003; Ward et al., 2007); showing less of an impact related to the warmer environment. Research suggests that vines, particularly of non-indigenous origin show larger and more sustained growth responses to increased CO_2 compared to trees (Hattenschwiler et al., 2003, Hickman et al., 2010). All climate modeling predicts an increase in ambient levels of CO_2.

Since the native range of these species is in similar climate zones, warming of temperatures will not expand the range in the Lake Simcoe watershed. However, the range of potential spread may increase northward into areas now not currently inhabited. Since the vine species are already in the watershed effective control of growth and further infestation is essential. Climate warming and increased CO_2 levels could enhance the growth of these plants in the region.

A variety of eradication and management options have been implemented to control the invasive spread of DSV. As with other highly invasive plant species the most effective control is the quick removal and disposal of the entire plant before the area is infested. The root mass must be dug out completely or re-growth will occur

Once an area has a large density of plants that make manual digging out of individual plants either cost prohibitive or impossible, the most effective control to date is the repeated application of systemic herbicides (Lawlor, 2006). Repeated application of herbicide (Glyphosate) to actively growing plants, either through cut stem or foliar spray is necessary. Flowering must have begun, but without seed formation, as viable seeds may still be developed once the plant begins to set seed. In thick infestations mowing prior to spraying may be necessary, and repeated application over several years may be required.

Experimental control through scything and mowing of plants, removal of flower heads and roots at the Fletcher Wildlife Garden in Ottawa has shown some success but is only applicable for smaller infested areas. In this instance the areas that have been manually controlled in this manner have had some success with the re-colonization of other species. (Fletcher Wildlife Garden, 2006, Hanrahan, 2009). This has not proven effective for invasion control. Biological control of this plant is in the experimental stage and is many years from being a viable option for the control of these species.

Emerald Ash Borer (EAB)

The Emerald Ash Borer (EAB) (*Agrilus planipennis* Fairmaire) is an exotic wood-boring beetle. The larva of EAB feed on the phloem and outer sapwood of ash trees in the genus Fraxinus, disrupting the trees ability to transport water and nutrients, and ultimately leading to tree mortality. Mature adults burrow a D shaped exit tunnel through the bark of the tree, emerging from late May through August, with peak emergence in early July (Scianna and Logar 2004). Adults feed for 1 to 2 weeks on ash foliage prior to mating (Scianna and Logar 2004). Females lay eggs between layers of bark and in bark crevices; larvae hatch in about 20 days and tunnel into the tree to feed on the phloem and outer sapwood (Bauer *et al* 2004, OMNR 2007, Scianna and Logar 2004).

The native range of the EAB includes Japan, Taiwan, Korea, and eastern Russia. Emerald Ash Borers were first discovered in North America in the Detroit Michigan area in July 2002. By October 2002, EAB was confirmed to be in Windsor, Ontario (Haack *et al* 2002). As of September 16, 2010 the EAB has not been positively identified in the Lake Simcoe watershed (USDA 2010); however, the southern and eastern edge of the watershed, located in the Regional Municipalities of Durham and York, are currently listed as an “infested place” (CFIA 2010) due to outbreaks in the southern portions of these municipalities.

All five of the species of ash native to Canada are vulnerable to EAB (Haack et al. 2002), although blue ash (*Fraxinus quadrangulata*) appears to be more resistant to the borer than the other species (Anulewicz et al. 2007). Research conducted in Ohio has shown that mortality of mature ash trees in an infested area is as high as 98% 6 years after an infestation by the EAB (Knight et al. 2010). To date the EAB has killed tens of millions of ash trees in North America, and threatens to kill all of the more than 1 billion ash trees in Ontario (Canadian Forest Service 2005), and estimated 7.5 billion ash trees on the continent (USDA Forest Service 2010). The EAB poses a major economic and environmental threat to urban and forested areas in Ontario.

There is very little literature on the potential impacts of climate change on the EAB and its distribution in Ontario. Investigations into the insect’s thermal requirements are ongoing (Lyons and Jones 2005). It appears unlikely that climate change will slow the spread of EAB in the Lake Simcoe watershed. Extreme cold temperatures can cause substantial mortality of EAB larva (Sobek et al. 2009). Preliminary research has shown that very little mortality occurs in EAB larvae until they freeze (Sobek et al 2009). The reported average freezing points of EAB larvae range from -30.6°C in Ontario, -25 °C in Minnesota. The warmer winters predicted in the Lake Simcoe watershed (Canadian Global Climate Model (CGCM2) - A2 Scenario, Colombo et al., 2007) will result in fewer extreme cold periods, thus potentially reducing the EAB larval mortality rates due to extreme cold.

Since EAB’s establishment in North America, major efforts have been underway to increase understanding of the ash borer’s biology and climatic requirements. In China the EAB typically produces one generation per year, overwintering as prepupae and emerging in the spring and summer (Haack *et al.* 2002, Poland and McCullough 2006). However, it has been observed that some EABs overwinter as young larvae, not prepupae, and therefore require a second year of development (Cappaert et al. 2005 and Siegert et al. 2005). Some have suggested that this is the result of colder climates (Haack *et al.* 2002), but others believe this may be the result of host resistance and quality (Poland and McCullough 2006). If climate is a factor in the number of seasons required to emerge as an adult, warmer winters may mean that greater number of EAB are able to produce a generation in one year.

Several approaches to stopping the spread of EAB have been implemented. Implementing a Quarantine has been the first line of defence. Movement of ash materials and firewood in pose the greatest risk of spreading EAB to uninfested areas. The Canadian Food Inspection Agency uses a Ministerial Order to regulate the movement of trees, nursery stock, logs, lumber, wood packaging, wood or bark, woodchips and bark chips of the Ash out of an infested area without prior permission (CFIA 2010). After the outbreak was first discovered in the Windsor area in 2002, CFIA removed more than 85,000 trees to create a 10 km wide and 30 km long (from Lake Erie to Lake St. Clair) corridor meant to act like a firebreak and prevent the beetle from spreading. However, given the beetles ability to fly such long distances, this did not prevent the beetle from spreading beyond the corridor, and such attempts have not been made since (Taylor 2010). While CFIA still carefully regulates infested areas, other treatments are being investigated and implemented.

Currently there is only one product available in Ontario for the control of EAB (Canadian Forest Service 2009). TreeAzin Systemic Insecticide has been registered by Health Canada's Pest Management Regulatory Agency in 2010. This insecticide inhibits emerald ash borer larval development, and prevents adult emergence if used preventatively or at the onset of an outbreak providing protection for two years with a single application.

Eurasian water milfoil (EWM)

Eurasian water milfoil (*Myriophyllum spicatum* L.) is a non-native, submersed, aquatic macrophyte optimally found in water depths of 1-3m (Aiken et al. 1979). The feather-like leaves are found in whirls of four, with 14-24 pairs of filiform divisions (Aiken et al. 1979). The number of filiform divisions is often used to distinguish *M. spicatum* from its native cousin, *M. sibericum* (Northern water milfoil), where the number of divisions tends to be less than 11 (Aiken et al. 1979). The plant is able to reproduce via the production of seeds, but the most dominant form of reproduction occurs asexually via shoot fragmentation and stolon production (Madsen and Smith 1997).

Since its widespread establishment in North America in the 1960s, EWM has become one of the most invasive exotic macrophytes introduced to the continent (Smith and Barko, 1990). This invasive-exotic species has spread throughout Ontario, British Columbia, Quebec and 45 of the 50 American states and has become a nuisance species that has re-shaped many aquatic communities (Smith and Barko, 1990). Locally, it is distributed throughout inland lakes and rivers in southern and central Ontario including the Trent Severn Waterway, Rideau Canal Waterway and many coastal wetlands and tributaries of Lake Ontario, Lake Huron, Lake Erie and Lake St. Clair (Trebitz and Taylor, 2007; OFAH, 2010). It is also present throughout the Lake Simcoe watershed including abundant populations in Kempenfelt Bay, Cook's Bay and the Eastern Shoreline of Lake Simcoe (OFAH, 2010). The plant is able to propagate early in the growing season across a wide range of water depth, clarity, and nutrient conditions (Aiken *et al.*, 1979).

Nuisance populations of EWM often develop dense monotypic stands, of which densities greater than 250 stems/m^2 are possible (Creed and Sheldon 1995). These monotypic stands are also associated with decreases in macroinvertebrate richness and diversity, dissolved oxygen, nutrient availability, flow disruption and adverse impacts to human activities such as navigation and recreation (Smith and Barko 1990; Aiken et al 1979).

Ultimately, it appears that factors such as sediment type and nutrient richness are what limit its dispersal (Barko and Smart 1986; Wang et al. 2008). EWM is typically dependent on finely textured inorganic sediments, and in many cases is limited to growth in areas with higher amounts of human activity, sediment deposition or groundwater infiltration (Lillie and Barko 1989; Smith and Barko, 1990).

Hybridization and genotypic variation within *Myriophyllum* genus has recently been identified across the Midwest United States and Ontario (Moody and Les 2002; Thum and Lennon 2006; Sturtevant et al. 2010). Locally in Ontario, hybridization between EWM and a native congener, northern watermilfoil has been identified in several of the Kawartha Lakes in 2009 (Dr. Ryan Thum, Personal Communication). It is unknown if different ecotypes or hybrids of EWM will lead to range expansion, increased invasiveness through heterosis, extirpation of its parent species through introgression or resiliency to herbivory or various forms of management (Moody and Les 2002).

Millions of dollars are spent yearly through management techniques in attempts to suppress and eradicate nuisance populations of *M. spicatum* across North America (Sheldon and Creed 1995; Pimentel 2005). Management techniques used to control nuisance populations include herbicide application, mechanical harvesting and biological control using a native herbivorous weevil, *Euhrychiopsis lecontei* (Newman 2004).

Harvesting of aquatic plants has been practiced for many years. The use of cutters, mechanical harvesters, dredgers, rakes, conveyors and pulling by hand often only accounts for short-term decreases in plant biomass, vast amounts of physical labour and can cause long term negative impacts to the biotic lake community (Aiken et al. 1979; Smith and Barko 1990; Unmuth et al. 1998). Practices such as raking and hand pulling often carried out by cottage/lakefront owners for swimming and navigational purposes come at a high labour cost. Although short-term removal of plants is noticeable, the potential for re-growth through fragmentation often occurs (Smith and Barko 1990; Unmuth et al. 1998).

The vulnerability of macrophyte communities to climate change is considered to be more apparent in shallow freshwater lakes and ponds in comparison larger lake systems (Carpenter et al 1992; McKee et al 2002). Increases in temperature are expected to impact growth capacity and nutrient availability within macrophyte communities, subsequently altering species diversity to favour competitive, exotic species (McKee et al 2002; Chambers et al 2008). This is often perceived to occur with range expansion of aggressive macrophyte species from warmer climates into regions with previously cooler climates (McKee et al 2002). The addition of anthropogenic disturbance to changes in climate may disrupt macrophytes on an individual or community level to provide invasive species, such as EWM, with a competitive advantage in such disrupted plant communities (Perkins et al 2010; Roley and Newman 2008).

Kudzu vine

Kudzu vine (*Pueraria lobata*) is a member of the pea family (*Fabaceae*) and is an aggressive, semi-woody, perennial vine. Its leaves are compound and alternate, with three leaflets, each 7-25 cm long. It produces an attractive spike of pink flowers (OFAH 2010). The vine has the potential to grow up to 30 cm a day, and up to 30 metres in one growing season under ideal conditions (OFAH 2010). It establishes large colonies through vegetative reproduction, forming multiple shoots from the same root system, and roots where the stem contacts the soil. Seed production does occur, but they have poor viability and low germination rates (Forseth and Innis 2004). Kudzu was first introduced in North America at the 1876 Centennial Exhibition in Philadelphia (GLU 2010), but the only confirmed occurrence (2009) of the species in Canada is near Leamington, Ontario (OFAH 2010).

Kudzu is native to eastern Asia where it grows in both lowland and upland mountainous habitats. The natural habitat is broad-leaved or mixed forest and forest edge; however, it also grows well in disturbed and altered habitats including road and rail embankments, pasture and agricultural or forest plantations (SE-EPPC 2010). The canopy of kudzu can reach depths of up to 2.5m (Forseth and Innis 2004) and thus its success comes from its ability to completely outcompete surrounding plants for light. The extreme rates of biomass accumulation are not limited to just the above ground parts. Kudzu roots have been shown to penetrate soils at a rate of 3 cm/day (Blaustein 2001) resulting in as much as 50% of total plant biomass being found below ground (Forseth and Innis 2004). With such large root biomass, it is able to fix significant amounts of nitrogen which allow it to colonize areas with poor soils (Forseth and Innis 2004).

Kudzu has maximum growth when winter temperatures are between 4-16 °C, summer temperatures exceed 26°C, and winter soil temperatures remain above -30°C (SE-EPPC, 2010). The plant has a reported hardiness zone of 7 (EPPO, 2007), however it has also been documented as hardy to zone 5 (CFIA, 2010). However, in far less favourable conditions, kudzu is viable and able to continue growth (Coiner 2007). Frost kills the semi-woody above ground vegetation, however the roots are extremely hardy and can survive colder temperatures. Although die off during winter will slow the growth of the plant, it will not eradicate it. Isolated populations in Massachusetts and Connecticut produce 10 to 15 m of annual stem growth although the stems are killed back to the overwintering crown each year (Sasek and Strain, 1990).

Kudzu has been spreading from the US southeast States northward as well as westward. Using climate modeling Jarnevitch and Stohlgren (2009) have modeled a northward trend of distribution in the northeast. The Great Lakes and St. Lawrence River have provided some physical separation from the most northerly affected States, however, it has shown the capacity to breach this divide as seen by the establishment of the colony on the north shore of Lake Erie in 2009. If this plant becomes established on the Ontario side of the Great Lakes, the climate warming scenario for the region will favour a more aggressive, vigorous growth season for the vine, with the added capacity to expand its range northward. The increase in seasonal winter and summer temperatures, combined with expected increases in atmospheric CO_2 will favour vigourous growth and fewer incidents of frost kill, allowing for a longer growing season.

Once established, Kudzu is extremely hard to eradicate. The entire plant must be removed as roots can overwinter. In southern areas, grazing, burning, mechanical removal and herbicides have all been tested. Experimental biological control is in the beginning stages (EPPO, 2007). So far control of this vine in the USA has failed and it continues to advance. Herbicidal application for 4-10 year period has proven effective in some cases (Global Invasive Species Database, 2010). It may be easier to eradicate this plant from more northerly regions as it cannot advance as quickly and will experience leaf and stem mortality in winter, however, once established aggressive management is a necessity. As with most of the invasive plants, early intervention is essential as plants established for a period of extended time are extremely difficult to eradicate.

1000 Cankers Disease (TCD)

In the last decade, Black Walnut die-offs in the western Unites States (Washington, Oregon, California, Idaho, Utah, Colorado, and New Mexico) have raised concerns surrounding an emerging disease affecting these trees: 1000 Cankers Disease (Tisserat *et al* 2009). This Walnut specific disease is caused by a new fungus species acting in an insect-fungus complex (Freeland *et al* 2009), to which the Black Walnut is especially susceptible. The vector, walnut twig beetle (*Pityophthorus juglandis*), carries the fungus (*Geosmithia morbida*, proposed scientific name) on its wing covers and deposits the fungus under the bark of the tree during the formation of reproductive galleries or tunnels (Tisserat and Cranshaw 2010). The fungus subsequently forms numerous cankers under the bark which girdle the branch or stem and leads to tree death (Tisserat and Cranshaw 2010).

The beetle is native to the United States, with its original range including New Mexico, Arizona, and Chihuahua, Mexico (Newton and Fowler 2009). They were not originally considered a pest, and as with most twig beetles their impacts were limited to "natural pruning" of the trees (Tisserat and Cranshaw 2010). It is only since the creation of the beetle-fungus complex that the walnut twig beetle has become a pest and threat to the walnut trees. While the walnut twig beetle may at one time have travelled independently of the fungus, the beetle-fungus complex are now consistently found together (Newton and Fowler 2009).

Identification of trees infected with TCD is complicated by the lack of external signs of the cankers. In order to see the cankers, a layer of bark must be removed from the suspect branch or stem. An additional sign is the presence of small exit holes left by the adult beetles when leaving the feeding galleries. If present, the bark around these holes can be peeled back to reveal the underlying galleries. While the walnut twig beetle may at one time have travelled independently of the fungus, the beetle-fungus complex are now consistently found together and so the presence of exit holes and galleries indicated the presence of the disease.

Additional outward signs from leaves and branches may be visible, but are only expressed after 5-10 years of infection. These symptoms include yellow flagging of smaller branches, often in the upper crown, thinning in the upper crown, and die-off of branches showing yellow flagging. In later stages, leaves may show extensive wilting. Death within 2-3 years of these symptoms becoming visible is common in the Rocky Mountain States.

While Black walnut is not native to the western states, it is planted extensively in both urban areas and plantations. The first record of large Black walnut die-off occurred in 2001 in the Espanola Valley of New Mexico. Other incidents of die-off may have happened earlier, such as in Utah in the early 1990's, but these moralities were attributed to drought condition and the presence of TCD was not determined. Other states with confirmed beetle populations and Black walnut mortality include Idaho (beetle presence confirmed 2003), Colorado (beetle presence confirmed 2004, but die-offs occurring since 2001), Washington (2008). The first case of TCD within the native range of Black walnut was found in July 2010, in Knoxville, Tennessee.

In eastern United States and Ontario, Black walnut is a native species present in natural plantings, landscaping, and plantations. Within this range, the species grows well in areas with deep, well drained soils. In native plantings, the species tend to be in small groupings or single individuals. The nuts produced by the species allow for animal dispersal of the seeds and thus their ability to colonize new locations at extended distances from the source tree. The susceptibility of native plantings of Black Walnut to TCD is undetermined because the disease has yet to have significant presence within the native range of Black walnut. However, in the western United States, even healthy individuals were susceptible to the disease. The patchiness of the specie in its native range may slow the transmission of the disease; however, plantations do exist in Ontario and these areas are particularly prone to disease outbreaks.

The distribution of the disease depends on the ability of the walnut twig beetle to reach new areas and to successfully overwinter in these new areas. Possible pathways for the transmission of the beetle have been proposed and discussed by Newton and Fowler (2009) and includes: timber, firewood, wood packaging materials, nursery stock, scion wood for grafting, nuts, and natural spread. Of these, only nuts has been discounted due to the apparent inability of the disease to transfer the fungus to the nuts produced by an infected tree. Beetles have been found to travel in timber and firewood products, both of which travel extensively in the United States. The fungus has been found in the cutting or off shoots used to propagate the species in nurseries.

The native range of the walnut twig beetle is very different climatically from Ontario: average minimum temperatures range from -12 to -1 (Plant hardiness zones 8 and 9, figure 2). However, the presence of the beetle-fungus disease in Colorado (zones 3-5) is cause for concern of its ability to successfully overwinter in Ontario and the Lake Simcoe watershed. In California, the beetle undergoes 2-3 generations per season and the larvae may be able to overwinter in these environments. In contrast, only 2 generations occur in Colorado with only the adults able to overwinter within bark cavities.

Regardless, in both locations the disease has been able to cause extensive mortality of Black walnut.

Control of TCD once established in a tree has not been successful. The beetle is susceptible to beetle insecticides; however, the beetle is active for long periods through the season (April-October) and is present in potentially many different locations within the tree, and thus control is difficult. The disease does require high beetle densities in order to kill the tree which may indicate that early detection is the key to control; however, as noted earlier, exterior symptoms are difficult to see and only appear years after infection.

Recommendations

There are numerous invasive species present in the Lake Simcoe watershed (see Table 1) and their date of introduction varies from early settlement time to modern day. The factors that are responsible for new introductions will vary by species, but we can learn from past experiences. Many of the plant species now known to be invasive were intentionally introduced by settlers for landscaping and horticultural reasons (i.e. Buckthorn used for hedges; Purple loosestrife as a garden plant). In spite of this, it is still possible to purchase invasive plants at garden centers for use in personal gardens.

The five species that were discussed in this report demonstrate the difficulty that managers face when invasive species populations become established and also the challenges of predicting the factors that are responsible for their spread. In some instances, a changing climate will facilitate the expansion of populations (i.e. Kudzu and TCD), but for others the extent to which we disturb and degrade our local communities will play a more significant role (DSV, EWM). Thus restoration of degraded ecosystems and mitigation of disturbance factors must play a significant role in any watershed management decisions that are focused upon invasive species.

In order to achieve the goal of no new introductions, as stated in the *Lake Simcoe and its Watershed* report, there will have to be a collective effort from all agencies, organizations, and the public to ensure rapid identification and reporting of new invasive species. With climate change expected to increase the success of invasive introductions, through warmer climates allowing overwintering, increased reproductive success, and existing range expansion (Walther *et al.* 2009), reporting new species before they become established will be critical to mitigating their impacts.

The Ontario Ministry of Natural Resources and Ontario Federation of Anglers and Hunters share the responsibility for the provincial invasive species monitoring program and database. Other professional organizations that are working in the watershed (Lake Simcoe Region Conservation Authority, Canadian Wildlife Service, Parks Canada) will also be recording their observations, as will local citizen and lake association groups. However, the management of invasive species is often very reactive in that it is only once the species has been identified that action takes place. To ensure that there are no new introductions, new preventative measures are needed. Recently the Invasives Tracking System (www.comap.ca/its) has come online for the Lake Simcoe Watershed and represents an excellent reporting system to identify invasives in the region. However, its success is still dependent upon public and agency participation. For example, despite its presence in the Lake and throughout the Trent-Severn Waterway, EWM does not have any reported sightings on the tracking system.

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