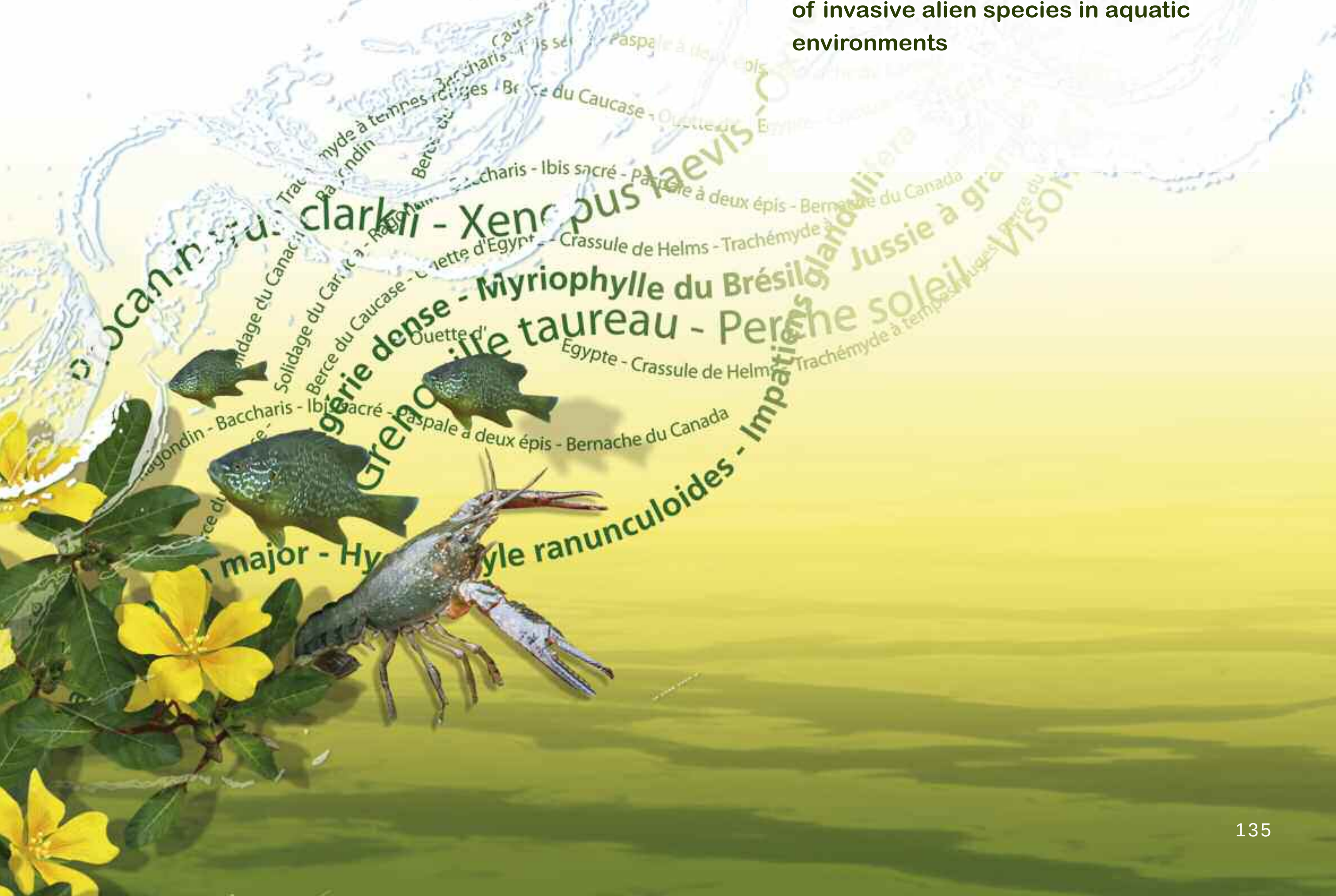
Management techniques for invasive alien species in aquatic environments

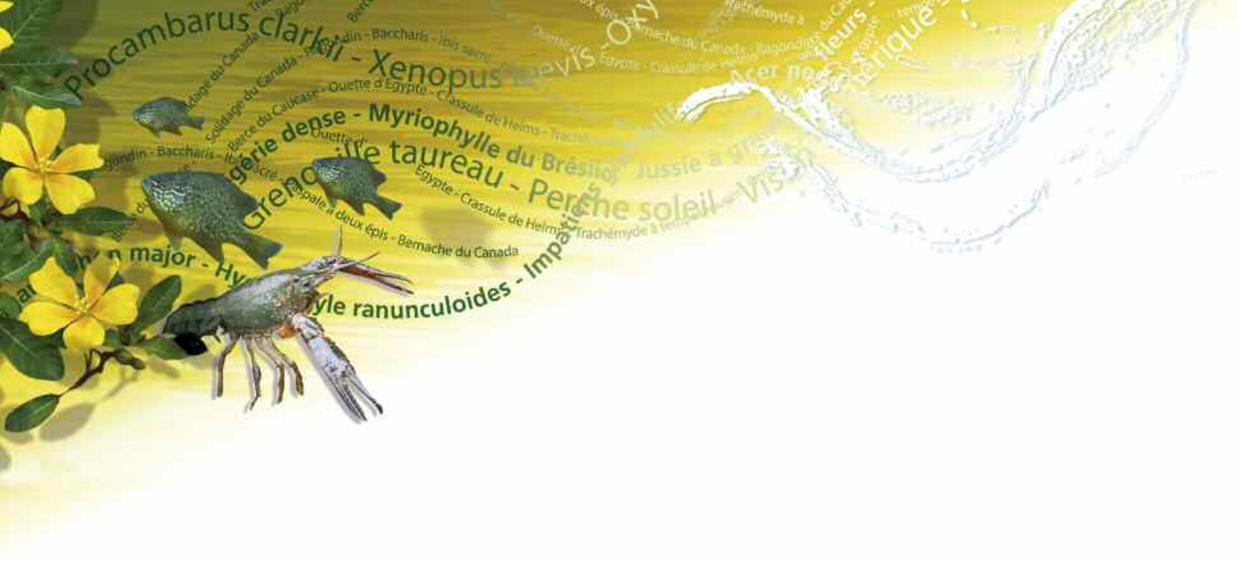
5

This chapter was drafted by:
Alain Dutartre (independent expert, formerly at Irstea)
and Emmanuelle Sarat (IUCN French committee).

The section titled "Organic transformation of invasive alien plants" was written by Hélène Gervais and Dorine Vial (Conservatory for natural areas for the Centre region).

With the collaboration of:
Jean-Patrice Damien (Brière regional nature park),
Nicolas Pipet (Sèvre-Niortaise basin interdepartmental institution),
Nicolas Poulet (Onema),
Jean-Philippe Reygrobellet (Board for balanced management of the Gardons basin),
Yohann Soubeyran (IUCN French committee).





Intervention guidelines step-by-step

Putting knowledge to work

The objective here is to work toward the actual intervention, on the local level, by gathering the available or accessible data on the situation requiring management, in order to make rational technical decisions. The local management approach (see Chapter 4), by collecting on-site information concerning the IASs and the needs of people, should facilitate these decisions.

■ Learning about the environment and how it is used

Site characteristics

Ideally, this phase of the approach should be used to collect all the available information on the environment(s) concerned by the intervention, e.g.:

- dimensions, surface areas, water depth(s), difference between high and low water levels;
- hydrological characteristics (flood and low-flow levels), development work and regulation of water levels;
- connections with other aquatic environments;
- plant and animal communities;
- types of banks and riparian vegetation;
- site accessibility;
- available or necessary equipment;
- regulations governing the site(s) and the planned work.

It is generally easy to collect this information, however it does not always exist in forms that are of immediate use for the analysis prior to setting up the intervention. In some cases, the information has already been collated in existing documents, however it is necessary to check the validity of the information.

Uses and the people using the environment

Effort should also be made to obtain the best possible information on how the site or the area are used. Uses may be defined in quantitative terms (energy production, irrigation, mitigation of low flows and of flooding in rivers, etc.) or qualitative terms (drinking water, swimming, hunting and fishing, etc.). They may take place in the specified environment or throughout the river basin. In the latter case, they may create functional limitations in the specific environment (Dutartre, 2002).

An analysis of uses on the site should indicate the types of uses, but also their geographic location and timing, the relative intensities of use and, if possible, their level of compatibility. Concerning the last point, many uses often take place in the same environment (see Figure 52), but the issues of how the available resources are shared and the interactions between activities are not always correctly assessed.

Figure 52

© A.Dutartre, Irstea

Dense beds of amphibious species, such as water primrose, can hinder the travel of boats in lakes.

For safety reasons, certain uses have for many years been regulated by the creation of zones (e.g. lines of buoys in a lake) or other limitations in order to reduce or eliminate the risks of accident. Examples are swimming areas cordoned off from boating sites, other areas where boating is forbidden, etc. (see Figure 53). These rules, intended strictly to ensure the safety of users, take into account only the physical characteristics of the environment (depth, etc.) and, in some cases, bank characteristics, but do not necessarily acknowledge the implications for environmental management.

Figure 53

© A.Dutartre, Irstea

These beds of parrot-feather watermilfoil block all access to parts of the pond where boating has already been prohibited. (Léon Pond, Landes department).

In other cases, contradictions may exist. For example, the presence of dense beds of submergent plants close to the surface may be highly troublesome for certain water sports, but very favourable for fish. Should efforts be made to eliminate or regulate the plants simply to benefit the water sports? At the end of the 1980s, the rapid colonisation of Blanc Pond (Landes department) by curly waterweed, in the form of dense beds covering approximately 100 out of the 180 hectares of the pond) elicited negative reactions on the part of all users, including anglers (see Figure 54 on the next page). A few years later, the reactions of the anglers had become more nuanced because it had been discovered that the plants enabled the development of a large perch population (*Perca fluviatilis*) thanks to the shelter provided to the alevins by the dense beds. Since that time, regular management work has been carried out on a part of the beds each year to enable boating activities (see the management report, vol. 2, page 23) and the pond continues to attract numerous anglers. Similar reactions of anglers confronted with such colonisations have been noted elsewhere.

Figure 54

Colonisation of Blanc Pond (Landes department) by curly waterweed (the dark section in the photo).

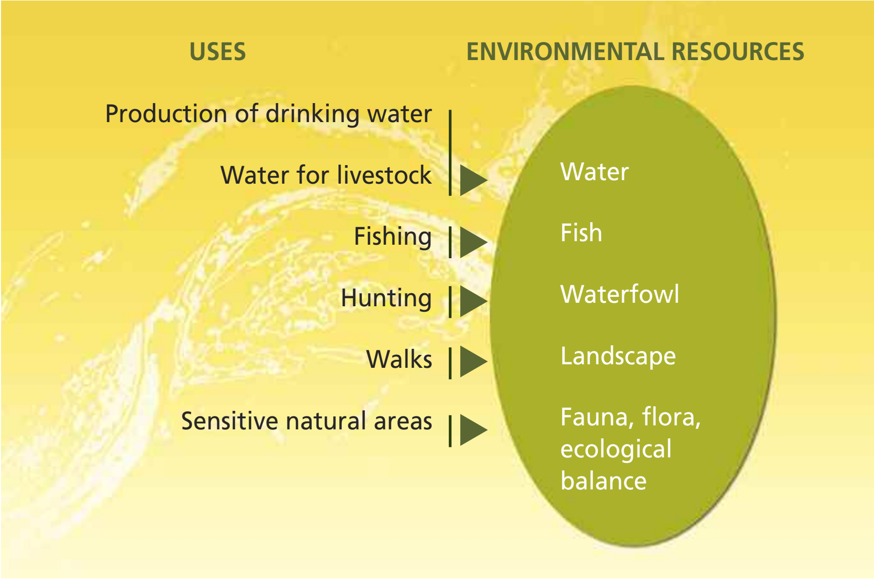
User groups, often organised in associations frequently try to highlight the importance of their particular use and in some cases neglect to mention the impact of that use on the environment. This lobby work increasingly focusses on the positive impacts that can result from growing tourism. These requests and defensive activities often lack any prior analysis of the issues and risks involved in the specific use, but they should be assessed taking into account the impacts they may cause for the environment. Their compatibility with the environment should also be examined. For example, if a large number of activities can co-exist in a large lake without endangering users, the same may not be true in a pond covering just a few hectares.

Similarly, certain uses in areas not conducive to that particular activity can rapidly oblige managers to undertake unforeseen interventions made necessary by the type of activity. For example, the creation of a swimming area in a lake or basin fed by a river, where algae or macrophytes may develop due to the high level of nutrients in the water, may turn out to be very costly in terms of maintenance. In some cases, an economic analysis of the maintenance work required could lead to the conclusion that a swimming pool may be a better choice that is less "natural", but for which maintenance is easier in terms of planning, cost estimates and implementation

These uses consume, in the widest sense of the term, environmental resources (see Figure 55 for an example of how a lake may be used and the corresponding resources). However, the availability, evolution and renewal capacity of the resources are frequently not assessed, which can lead to significant differences between the expectations of users and the mid to long-term capacity of the environment, and consequently to dissatisfaction on the part of the users.

An analysis may therefore need to set priorities among uses (determining the main use and the remaining secondary uses), thus facilitating later decisions on intervention.

Figure 55



Uses and resources, the example of the Pen Mur millpond in the town of Muzillac (Morbihan department). According to Dutartre et al.*, 1997.*

Assess disturbances and damage

■ Disturbances and their causes, an analysis of the expressed problems

Another element in this initial approach is to identify the disturbances noted by users and/or managers. This consists of a subjective assessment of a problem for one or more uses caused by changes in one or more parameters in the environment, where the disturbance is defined in terms of these uses. The development of aquatic plants covering large surface areas or in places used by people, or the excessive quantities of bird droppings in lakes or on lawns in parks may be said to be disturbances if they hinder those uses.

A disturbance is therefore defined with respect to one or more uses and an analysis of the situation must attempt to reduce the subjectivity inherent in most definitions and develop the level of objectivity required to set up suitable environmental management. This analysis must take into account the uses and the management objectives, while respecting the functional consistency of the environment (Dutartre, 2002).

Disturbances are therefore not the same thing as the damage caused by IASs, which have to do with the impacts incurred by these species (see Chapter 1). Competition with native communities, uniform landscapes, local regression of biodiversity or predation by the invasive fauna are the damages most frequently observed on a site or in an area.

■ A necessary assessment of the damage

Ideally, an assessment of the damage should be carried out before launching an intervention, but the task is highly complex (see Box 21 on the next page). This is because the data on any damage caused by the invasive species are often not available on the site or in the area in question. In addition, managers generally do not plan on conducting assessments or analyses on the damage caused by IASs and to date very few studies to quantify the damage have been funded.

The historical data on species distribution ranges and ecosystem functioning are required to detect the mechanisms and processes involved in a colonisation and in the resulting spread of populations and disturbances. For comparative studies, undisturbed control sites are not always available and, finally, indisputable assessment protocols designed to detect and quantify damages must still be established (Haury *et al.*, 2010).

The history of a data sheet used to provide management advice

The continuous improvements in knowledge on species and on management techniques, in the training of managers, in the dissemination of information, etc. mean that it is now possible to offer solid information that can be easily transmitted by the internet. This is a relatively recent development, made possible by constant progress in the information and communication technologies. Three decades ago, prior to the arrival of the new techniques, the situation was quite different.

The first proliferations of aquatic plants, either native or alien, that were seen as sufficiently serious to provoke a reaction from stakeholders and managers date back, as far as we are aware, to the 1970s. In that no communication networks had been set up at that time, the requests were sent to various institutional contacts, such as the Departmental agricultural agencies and the Water agencies. These institutions did not have the necessary knowledge bases or human resources, consequently they transmitted the requests to other organisations seen as more likely to provide answers, such as INRA and Cemagref where research teams worked on aquatic environments, for example in Thonon-les-Bains (INRA) or in Lyon and Bordeaux (Cemagref).

Though few in number initially, these requests resulted in a joint assessment by INRA and Cemagref on a lake in the Charente-Maritime department (Dutartre *et al.*, 1981). The number of requests grew rapidly in the 1980s, arriving from departmental services, managers and even private land owners. Most presented the problem in simple, even simplistic terms because the perceived difficulty appeared very simple, i.e. too many aquatic plants! The requests could most often be summed up in the following sentence, *"This plant is creating a problem, what is the most effective herbicide?"*. At that time, a number of herbicides had been approved for aquatic environments and they were seen as effective, inexpensive and relatively inoffensive for the environment. However, it very rapidly became clear that without further information on the local situation, it was impossible to propose a consistent, structured response, capable of limiting environmental damage and the risks of ineffectual action.

At that time, regular meetings were held from 1978 to 1994 by the *Aquatic plants* group, set up under the auspices of the Weed committee (COLUMA) that subsequently became the National association for plant protection and finally the French association for plant protection (AFPP). The group produced over 35 documents on the biology, ecology and management of aquatic plants, presented during either internal meetings or *Aquatic plants* sessions held during COLUMA conferences in 1990 and 1992 (Dutartre, 1994). Its primary contribution consisted of updating a book containing identification sheets for the main aquatic plants, a procedure for plant identification and information on management methods (Montégut, 1987). The group also collectively prepared an "assessment-aid data sheet" used to collect, partially in coded format, information of use in preparing management plans and in providing consistent answers to questions. The information recorded on the sheet included sizing dimensions and the local uses of the area in question, the types of plants considered responsible for the disturbances and any work already carried out against the plants.

The person or entity requesting assistance was instructed to fill out the sheet as completely as possible. It was then used as the basis for management advice that in some cases led to more in-depth discussions on the difficulties of managing aquatic plants. Over time, we noted that by sending the data sheet as a first step prior to providing advice, it was possible to regulate the volume of requests. A return rate of 50% fairly rapidly revealed the overly dramatic nature of some requests.

An analysis of approximately 50 data sheets and of a similar number of answers to a survey run by the group in 1988 (Dutartre, 1992) confirmed that part of the difficulties encountered were due a lack of knowledge on the ecological functioning of environments. Other problems were the direct result of efforts by managers to develop multiple uses of environments without taking into account the capacity of the environments to fulfil those requirements. Among the major missing pieces in the puzzle was taxonomic information on the species in question, which led to serious management errors when the species causing the disturbances was incorrectly identified.

Since that time, the data sheet, with a summary of the theoretical management approach (a very short presentation) and, later, digital copies of articles on plants and their management, has been put to regular use. In many cases, it initiated more in-depth and functional discussions than would have been possible by simply sending informational documents and thus contributed to the progressive constitution of the current network of people working on the management of aquatic plants, even though, in the beginning, requests concerned both native and alien species, a situation arising much more infrequently today.

The data sheet is now fairly old and less useful due to the large amounts of information available via the internet, but it can still be used for requests from people who are not at all familiar with the topic. It remains the first step in discussions that can significantly improve the advice provided and make clear to the requesting person that they are entitled to ask simple questions, but any serious response first requires diverse information on the environment, the species and the uses to which the environment is put.

The data sheet, which must be improved and updated, is available on the IBMA site (www.gt-ibma.eu) with other tools to assist management (worksite progression sheets, monitoring protocols, etc.).

■ Difficulties involved in assessments

Very roughly speaking, it may be possible to say that disturbances are more qualitative in nature (*"I am not satisfied"*) and damages more quantitative, but in fact the situation is more complicated. The impacts of a species on an activity can in fact be rapidly quantified when those impacts interfere with certain economic activities such as tourism. For example, what would be the economic losses to boating in the Marais Poitevin marshes if the spread of water primrose were not regulated (Pipet and Dutartre, 2014)?

On the other hand, it is not always easy to quantify the damage done to nature by IASs. Cost assessments of the damage, generally on the national and international scales, have been carried out over the past decade. One of the better known assessments is that of Pimentel. He calculated that the economic and environmental losses for the basin of the Great Lakes in North America represented 5.7 billion USD (Pimentel, 2005) and the total annual loss for the United States as a whole represented 120 billion USD (Pimentel *et al.*, 2005).

In Europe, the work by Kettunen *et al.* (2009) is regularly used as a reference point in calculating the annual costs of damage caused by IASs and the management work required to control them. The total amount often mentioned in European documents is 12.5 billion euros.

But in fact, Kettunen and his colleagues wrote that *"According to existing data the total costs of IAS in Europe are estimated to be at least 12.5 billion EUR per year (according to documented costs) and probably over 20 billion EUR (based on some extrapolation of costs) per year)".*

The difficulties in obtaining relatively precise numbers do not necessarily decrease in step with a drop in the size of the area being analysed. The costs of interventions are, theoretically, relatively easy to determine because they are often covered by public funds. But the participation of volunteers in a wide array of management operations, e.g. environmental monitoring, warning networks and intervention work, is often not included in the calculations of the "social costs" of IAS management. Even if this participation is most likely limited, it should be taken into account so that the total cost is as accurate as possible and provides a better overall description of IAS management.

Concerning the costs of damage, some may be calculated on the basis of the direct economic losses caused by IASs for certain human activities in the area under consideration. On the other hand, damage to biodiversity, in terms of species, living communities, habitats and the ecological functions of environments, is not specifically taken into account and therefore not included in assessments.

That is why study and work have, for a number of years, been put into determining the ecosystem services provided by environments in order to widen the assessments of IAS impacts (Amigues and Chevassus-au-Louis, 2011). The generally accepted definition of "ecosystem services" states that they are *"benefits that humans gain from ecosystems without doing anything to obtain those benefits"*. It is therefore necessary to establish reference values for ecosystems that can then be used to calculate the reduced benefits caused, for example, by IASs and include the results in the overall economic analyses. These analyses generally deal with major types of ecosystem, e.g. forest ecosystems, mountain ecosystems, marine and coastal ecosystems. Current work by the IUCN French committee has already produced a number of documents on the topic (see http://www.uicn.fr/-Outils-et-documents-.html). For example, a report on continental freshwater ecosystems has already been published (UICN France, 2014).

In 2010, following the Grenelle environmental meetings, a report on the economic assessment of the services provided by wetlands (Aoubid and Gaubert, 2010) presented data on these environments (including alluvial plains, marshes, peat bogs, estuaries, artificial lakes, ponds, littoral wetlands). In that the Grenelle agreement foresaw, for their preservation, the purchase of 20 000 hectares of wetlands by the Seaside and Lake Conservation Trust and the Water agencies by 2015, the assessment produced figures for an equivalent surface area. The report took into account a wide range of direct and indirect services provided by wetland ecosystems. The resulting economic assessments showed that the loss of 20 000 hectares of wetlands, i.e. the loss of the corresponding functions and benefits, would over a 50-year period incur costs of between 405 million and 1.4 billion euros. When compared with the cost of purchasing and maintaining 20 000 hectares of wetlands, i.e. 200 to 300 million euros over the same time span, the benefits of preserving wetlands are clear.

A French programme to assess ecosystems and ecosystem services (EFESE) was launched in 2013 under the responsibility of the Ecology ministry (http://www.developpement-durable.gouv.fr/Levaluation-francaise-des.html). The objective is to establish a multi-disciplinary network of researchers working on ecosystem services. Finally, in 2014, the Sustainable-development division of the Ecology ministry (CGDD) started a survey titled *The cost of invasive alien species in France* and the results should be published in 2015.

Consequently, this approach to ecosystem services is not yet available for assessments on IAS damages in aquatic environments. That being said, the information on costs and measurable impacts, gathered by managers and researchers from past interventions, is increasingly well organised and available in the databases constituted to support this approach to ecosystem services.

Finally, though the assessments in some cases are sufficient to justify interventions and obtain the necessary financing, in many other cases managers are at a loss to present data on economic justifications. Locally, an assessment of the disturbances or damages cannot begin until they have become "perceptible", i.e. when the complaints of stakeholders or observations on IAS impacts in the area managed have become sufficient to trip a reaction. The preventive efforts set up in application of the European regulation should contribute to improving knowledge on IASs and facilitate assessments of the damages caused.

Learning more about IASs causing disturbances and damage

Clear identification of species (taxonomy) is indispensable in precisely defining the management problem, but is still neglected in some cases. One of the main advantages of identifying a species deemed responsible for disturbances is to gain access to the specific knowledge available on its biology, ecology and the techniques used to regulate its population.

For example, a species may proliferate via cuttings (the case for water primrose and Hydrocharitaceae), another may prefer biotopes protected from the wind (e.g. water fern), uprooted plants deposited on a site too close to water may cause a new contamination, etc. Concerning fauna, a healthy, alien species may carry mortal diseases for native species (invasive crayfish and amphibians are well known examples), another may be strictly nocturnal and efforts to shoot certain animal species may simply cause the populations to disperse.

In-depth knowledge on the geographic distribution of species is also required prior to management operations. An objective of interventions is to limit the dispersal of the local populations. Detection of colonised sectors, of invasion fronts and monitoring of adjacent areas (e.g. over a river basin) are means to identify the priority areas for interventions, depending on the objectives of the manager. An assessment of the dispersal potential of the species and of any favourable, nearby biotopes can usefully complement the information on the colonised sectors and serve to refine the monitoring strategy for the area. A number of mapping tools already exist and others are now being developed by managers and various IAS work groups (see Chapter 6). They can be of assistance in prioritising interventions.

Finally, in some situations, interventions have already been carried out, often without the necessary precautions and prior analysis required to reduce implementation risks. The information on past interventions and their results should also be taken into account (Haury *et al.*, 2010).

Assessing the ecological issues of management interventions

In general, management interventions themselves have impacts on the ecosystems in which they are carried out, either specifically on certain species or living communities (fauna and flora) not targeted by the work, or more generally on the functioning of habitats. In that interventions can cover large areas and given the currently clear requirements in terms of biodiversity protection, the need to better assess the consequences of management work would appear manifest.

The objective is not to limit intervention possibilities through the systematic application of the precautionary principle, but to base management decisions on as much information as possible by first comparing the IAS damage to the impacts of management. This assessment must obviously be based on the available data pertaining to the biology and ecology of the targeted species and the previously observed impacts of the potential intervention techniques on the living communities not targeted and on the environments involved.

Information previously acquired from other cases on IAS expansion dynamics (areas covered, populations) is most useful for the assessment. In some situations, that may lead to intervention techniques deemed “violent” and even debatable, but that may be justified in light of the ecological issues..

For example, New Zealand pigmyweed (*Crassula helmsii*), a small plant growing near water and considered particularly invasive in continental France on the basis of information from the U.K., has been systematically monitored for several years. Following its observation in a pond on a site to the east of Donges (Loire-Atlantique department), it was decided in 2012 to fill in the pond in order to eliminate the plant from the site (Sauvé and Rascle, 2012). But to enable the departure of the amphibians living in the pond invaded by the New Zealand pigmyweed, filling occurred in two separate steps (2012 and 2013) and a substitute pond was created near the invaded pond. No New Zealand pigmyweed was observed during the inspection of the site in July 2014 (Matrat, personal pub.). Work to uproot the same species has been carried out in a pond in the Deux-Sèvres department (see the management project in volume 2, page 47, and Figure 56) and proved the usefulness of the technique.

Figure 56

© IIBSN

Manual uprooting of New Zealand pigmyweed in the Deux-Sèvres department.

In other cases, the situation may be seen as critical and though the impacts of the proposed work may be high, even very high, the work is nonetheless carried out to avoid further worsening of the situation. For example, the shallow Turc Pond (Landes department) was colonised for over ten years by large-flower water primrose, to the point that the plant had totally invaded two of the eight hectares, completely eliminating the other aquatic plants and disturbing local uses. In 1993, a floating platform equipped with a mechanical claw was used to remove 5 600 cubic metres of plants and sediment (Dutartre, 2004). This technique provoked significant “mechanical pollution” by suspending superficial, organic sediment in the water, which may be detrimental to fish populations. Unfortunately, no other techniques were available in the given context. The same problems were created recently in the Sologne Pond where water primrose was mechanically uprooted and sediment was dredged (see the management project in volume 2, page 63).

Certain management techniques can impact native animal populations. For example, shooting invasive alien birds or harvesting new water primrose in the spring can trouble native species present on the same site. Precautions must be taken, e.g. using guns equipped with silencers or subsonic ammunition, organising shooting campaigns outside of reproductive and nesting periods, etc. For amphibians, mistakes may be made when removing eggs or destroying metamorphosed animals, particularly if this work is done by private persons with no training. In order to avoid mistakes for all species, work is systematically carried out or supervised by qualified personnel (National agency for hunting and wildlife, personnel from national reserves, environmental-protection associations, local governments). It is also regulated (prefectoral lists of persons authorised to carry out work).

Defining management objectives

Defining management objectives is indispensable, but often neglected as if it could be dispensed with. Is the objective to reduce the space occupied by the species in the environment? Has a request been made to do so? What damage or disturbances must be limited or avoided? What is the targeted future status of the site or area? Confusion is regularly observed in many situations. The technique used to regulate the species is often seen as the objective and this ambiguity leads in many cases to imprecise interventions and consequently to unsatisfactory results.

In environments used for a single or small number of purposes, definition of objectives may take place fairly rapidly. The same is not necessarily true for environments used for many purposes, where negotiations between stakeholder representatives may be necessary to set valid objectives for the subsequent work. For example, an invasive alien plant may hinder some water sports, but contribute to the landscape and fishing, an alien bird may contribute to eutrophication of the environment, but have ornamental value for the general public, etc. In addition, a simple reduction in numbers or in surface areas colonised by a given species is not in itself an objective. An assessment of the disturbances or damages taking into account the targeted species must be the starting point in defining objectives for management interventions. Certain invasive species may continue to cause problems, e.g. those carrying and transmitting pathogens such as alien crayfish and American bullfrogs.

However, in order to produce the objectives most likely to result in the management work best suited to the situation, the negotiations must be based on a complete description of the site, the observed damages and disturbances, the local uses and the relations between those uses. There should also be a discussion phase between stakeholders, even if it may appear to delay the actual work. The purpose is to define objectives shared by the stakeholders, that are realistic given the colonisation dynamics, the available technical resources and budgetary limitations.

Defining an intervention programme

Local interventions that are inexpensive because limited in scope (space and/or time) can be carried out directly by the local managers, generally without needing to involve higher echelons. However, as in all cases and to the degree possible, they must be suited to the targeted situation (type of IAS, the environment, the human needs, etc.). In some cases, local interventions, carried out with unsuitable or insufficient means (human, material, financial) or without the precautions required by the given species, have resulted in fiascos that occasionally have produced situations even worse and more difficult to manage than the original situation.

Figure 57

Adaptive management. According to Tu et al.*, 2001.*

When the management project requires successive interventions, as is often the case, they must be organised in a precise programme stipulating the objectives, the sites, the techniques used to collect and dispose of the plants or animals removed from the site, the schedule, etc. From the start, this programme should be designed in an adaptive framework (Tu *et al.*, 2001) (see Figure 57), including regular re-assessments of the situation on the basis of the results obtained by the interventions and recent scientific information likely to modify the objectives or the intervention techniques. This regular review of the situation ensures the best possible management results because it requires constant vigilance, it theoretically avoids falling into a routine lacking any analysis and it re-assesses the operational management conditions on the basis of the results obtained.

For many years, an intervention programme has been one of the key documents among the application forms for financial aid that local managers must submit to public authorities such as the Water agencies or to local governments providing such aid. The projected intervention programmes enable the funding entities to better assess the issues and determine whether the planned action will be effective. Intervention programmes may also include proposals for experiments on technical aspects for which uncertainty subsists or intervention techniques designed for special types of sites.

For example, the proposal for the management plan for lakes and ponds in the Landes department, prepared for the Géolandes management board (Dutartre *et al.*, 1989), included tests on several very precise sites, trials using herbicides (at a time when certain products were still permitted for aquatic environments) and a section explaining the value of subsequent monitoring. The great amount of information now available on most IASs makes it much easier to draft intervention programmes and a number of current and past programmes can also be of assistance in preparing and drafting the document.

That being said, in many cases, it remains very difficult to provide sufficiently precise information on future interventions, even from one year to the next, because various random factors can affect sites and the IAS populations. Concerning flora, plant development is influenced by winter and spring weather. For example, in south-west France, shifts of a full month in the flowering and maximum biomass production of water primrose have been observed, depending on spring temperatures and sunlight. In rivers, winter and spring flood regimes can have direct and strong effects on hydrophyte development.

In the adaptive management system mentioned above, the indispensable monitoring required to assess the effectiveness of interventions and improvements in situations must also be an integral part of a mid and long-term management plan and consequently a key element in provisional budgets. Depending on the local situation, monitoring can address the targeted IAS populations and the non-targeted living communities, any disturbances caused by the intervention, the risks of new invasions, etc.

Concerning fauna, species' development cycles are also linked to climatic conditions. For example, high spring temperatures bring forward the reproductive cycle of American bullfrogs and provoke early laying of eggs, and a mild fall encourages further activity of adults, whereas management work is generally programmed over a shorter time period. Heavy rainfall may inhibit nightly shooting campaigns and prevent access to the banks of ponds. Other unforeseeable events, such as disagreements with land owners on the proposed work (fishing and emptying of ponds, installation of trapping barriers) and conflicts concerning land use (management work inhibiting hunting and fishing), can also occur and complicate management.

These unforeseen events create difficulties for interventions, particularly concerning certain aspects of funding. Variability in the development of species can render some planned interventions totally useless or make necessary other forms of intervention that were not initially planned and cannot be carried out unless additional funding becomes available. As a result, continuous adjustments in planning are required, in turn leading to reorganisation of the human resources involved and, for fauna, modifications in the prefectoral authorisations for management operations.

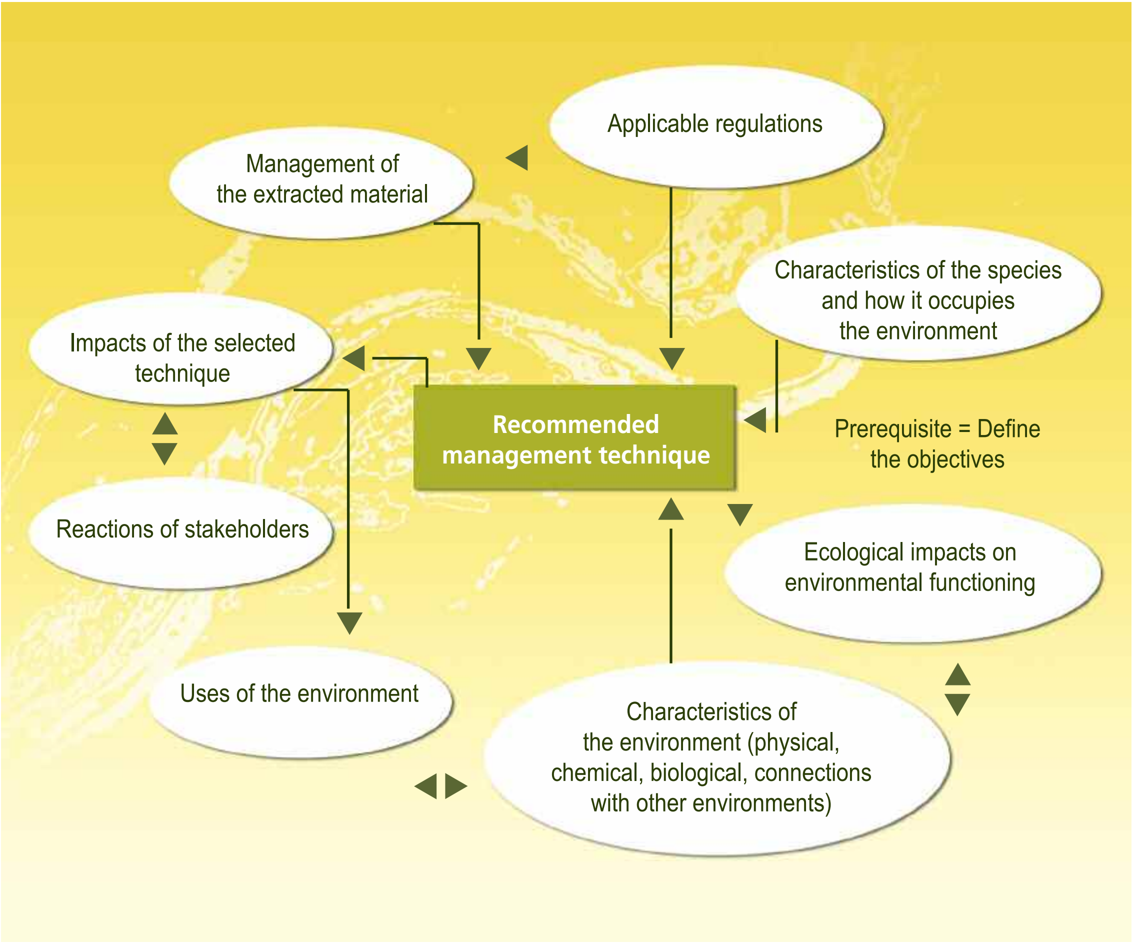
Selecting the intervention method

Intervention techniques should be analysed and selected depending on the previously defined objectives. Figure 58 presents the topics for analysis that can help in producing a rational choice. These elements include the available information on uses and disturbances, the species in question (biology, ecology), how it occupies the environment (distribution, colonised biotopes, etc.), the environment including its connexity with other environments where interventions may produce direct or indirect impacts (see Box 22 below), etc.

One indispensable rule is that "none of the available intervention techniques can be used for all situations". Each has a number of limitations that must be taken into account when making a selection. These limits to their application possibilities are now fairly well understood. In addition to guiding decisions, they must also be listed in the technical specifications for the work and discussed during the negotiations with the companies that will carry out the interventions.

The technical decision must then be analysed in light of the available human and financial resources to determine whether implementation is possible. The technical decision should be taken before examining the economic aspects because the overriding objective is to set up interventions suited to both the site and the species in order to obtain the best possible results. The point here is not to minimise or ignore the economic constraints that will come to bear even in the best of situations, but to target the most effective techniques in view of reaching the set objectives.

Figure 58



Topics for analysis in selecting intervention techniques. According to Dutartre et al.*, 1997; Dutartre, 2002.*

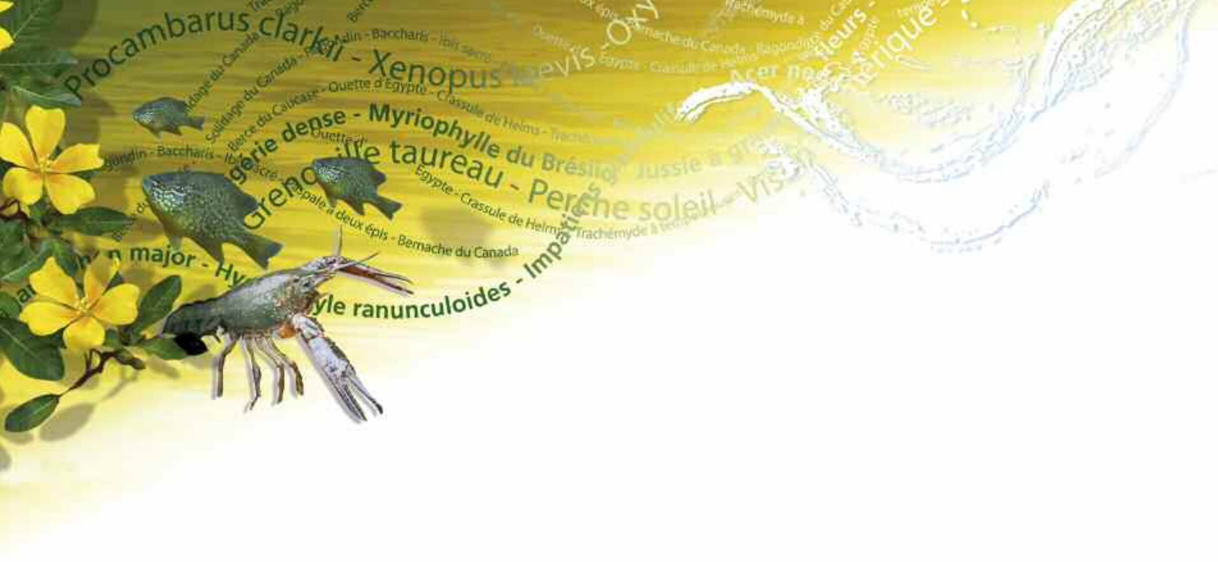
If the financial resources are insufficient or cannot be increased to the necessary level, it would be counter-productive in many cases to implement, for financial reasons, unsuitable technical methods that may cause unforeseen damage and not fulfil the set objectives. For example, a decision to use a given set of equipment simply because it is available, even though it is poorly suited to the species and the site in question, may result in severe failure and a worsening of the problems caused by the IAS. When funding is limited, it is always possible to prioritise intervention sites, targeting only the most important. But, similar to other fields of environmental management, the objective is rather to determine the “price” that can be assigned to the managed environment, i.e. the amount of money that society is ready to pay for the management in order to reach the set objectives.

Controlling the control method

Among the theoretical studies on management techniques, a common topic concerns the possibility of controlling the control method if the intervention produces unexpected consequences. It is not always possible to control the management techniques and it is worth putting thought into planning for the unexpected.

In terms of plant management, when manual or mechanical means are employed, the work can simply be stopped and the limits to the intervention are clearly defined. That was not the case when herbicides were used because spreading of the products, even when there was no wind or current in stagnant aquatic environments, could affect areas much larger (by a factor of 1.5 to 2) than the treated area. Finally, the use of biological-control agents cannot be confined and the organisms may progressively spread to the entire environment in which they were introduced, plus any other favourable, nearby or connected environments, if the targeted invasive species has dispersed significantly throughout the area. In addition, if the control agent shifts its consumption habits or pathological development in the area where it is introduced, it “betrays” the introducer and completely unforeseeable management difficulties may arise. One factor that differentiates manual/mechanical techniques from the others is the need to dispose of the waste produced in order to avoid any secondary dispersal.

Concerning fauna, the animals are generally trapped or shot. Trapping is a more precise management technique, but it is absolutely necessary to check that the means employed target the correct species and to assess their impact on non-targeted species. For example, the nets used to capture crayfish can also catch protected species such as eels (*Anguilla anguilla*) or European pond turtles (*Emys orbicularis*) (Poulet, 2014). Shooting is generally used for birds or amphibians. Though more selective, there are also more risks involved.



Panorama of management techniques

Before presenting the techniques used to manage invasive fauna and flora, it is certainly worthwhile to note that a "healthy" environment is generally thought to be less sensitive to invasions. A healthy environment cannot totally block the adaptation of one or more alien species, but it would appear increasingly clear to a wide array of researchers that a lack of degradation in an ecosystem means it retains a higher degree of resilience to biological invasions than a degraded ecosystem, with as a result lower management costs.

In addition, prior to a project, it is absolutely necessary to gather information from the relevant scientific and technical organisations (public agencies, National agency for water and aquatic environments (Onema), National agency for hunting and wildlife (ONCFS), universities, regional scientific councils on natural heritage, etc.) and to contact the State services processing applications (regional environmental directorates, departmental territorial agencies, etc.).

Panorama of management techniques for plant species

Management techniques for plant species can be divided into two groups, the first concerns means to prevent plant development and the second concerns work on the plants themselves.

Preventing plant development

Prevention consists of interventions that modify certain ecological characteristics of the biotope. These interventions fall under the heading of “habitat manipulation”. They may reduce or hinder the development of plants, native and/or alien. They can have an effect on water quality (nutrients, salinity), on the incident light, on water levels or on current velocities.

■ Reducing nutrients in aquatic environments

Aquatic macrophytes can feed either via their root systems or directly through the tissues of their stalks and leaves. It follows that the physical-chemical quality of water and sediment, in particular concerning the availability of nutrients, can directly influence the development of plant communities. The efforts undertaken over several decades to reduce eutrophication, consisting of work to limit the quantities of nutrients arriving from river basins, have effectively reduced the development of phytoplankton in many rivers and lakes (see Box 23 and Figure 59).

The results have not been as good for macrophytes, with the exception of floating plants that draw their nutrients from the water itself. This is because the links between nutrients and rooted plants are very complex and differ between species. The type of sediment can also play a role in the biomass production of submergent macrophytes (see, for example, Anderson and Kalff, 1988). In addition, an accumulation of organic sediment, which can occur in large rivers with slow to moderate currents and in all stagnant environments, can result in large quantities of nutrients. Given that there is often very little nutrient transfer between sediment and water, there is high potential for biomass development in macrophytes in biotopes where there are large quantities of organic sediment, even when the nutrient content in water has fallen considerably.

The development of submergent plants, both native and alien, observed for several years in a number of large rivers such as the Rhône and the Loire, are clear examples of these complex relations. Efforts against eutrophication have significantly reduced nutrient contents in water and consequently limited the development of phytoplankton, but have in the process increased the amount of light in the water and encouraged the growth of rooted macrophytes. The improvements in water quality can result in plant growth that is deemed excessive.

It should be noted that plants grew on the planet well before anthropogenic nutrient releases occurred in their ecosystems, consequently some are capable of colonising biotopes where nutrient contents are very low. The sediments most often colonised by certain submergent or amphibious invasive plants (Hydrocharitaceae and water primrose) are rich in organic matter (mud, peat). However, the same species can colonise biotopes that are apparently less favourable in terms of the available nutrients, but offer all the other environmental parameters required for their installation and continued development. For example, curly waterweed or large-flower water primrose can establish and maintain fairly dense stands in the relatively inorganic sand on the eastern banks of lakes and ponds along the Aquitaine coast. The simplistic relationships that are still regularly mentioned, e.g. between "water primrose and eutrophication" or "submergent Hydrocharitaceae and highly organic substrates", are not necessarily valid.

Though efforts to reduce nutrients in aquatic environments would still appear useful in order to pursue reductions in the ecological disturbances created by anthropogenic nutrient inputs, they cannot necessarily limit the development of macrophytes in many environments, except perhaps in special cases, e.g. the proliferation of floating plants in stagnant environments.

Box 23

Sediment treatments to inactivate nutrients or reduce organic matter

For several decades, various processes to block the phosphorous in sediment have been used in many lakes and ponds. In general, the water bodies were reservoirs used for drinking water in which phytoplankton had developed, causing problems for the potability treatment.

The products added to the water, either aluminium sulfate or iron(III) chloride, or a mix of the two, formed chemical compounds with the phosphorous that were thought to be stable. Another process used calcium nitrate to create an oxidised layer on the surface of the sediment and limit the release of phosphorous in the water.

The purpose of these treatments was not to reduce the macrophytes communities. And it turned out that the macrophyte communities, by blocking the even distribution of the products, reduced the effectiveness of the treatments. Furthermore, by increasing water transparency due to the flocculation of suspended particles, the treatments could even encourage the development of plants in the water bodies.

Over the years, other sediment-treatment products came into use. These products, theoretically non toxic because "natural" (consisting essentially calcium carbonate (Lithophyllum or coccolith chalk) or various hydrated aluminium silicates (zeolites, kaolinite), were proposed in great quantities to a large number of local governments by companies specialised in treatments for small water bodies.

Initially, the general objective was still to reduce the quantity of organic matter in sediment and to "reduce eutrophication" of the environment (Garnier-Zarli *et al.*, 1994), then, in step with the colonisations of plants in many water bodies, the sales pitch shifted to present them as "ecological" means to reduce sediment build-up and control (or even eliminate) macrophyte development. The next marketing phase was to incorporate in the mineral substrates a wide range of bacteria intended to enhance the effectiveness of the products that thus became "bioaddtives" and benefited from an aura of innovation, ecology and respect for the environment.

Given the large number of treatments carried out without any technical monitoring or assessments of their true effectiveness, it was decided to look at the results of their implementation throughout the entire country (Goubault de Brugière and Dutartre, 1997). The study revealed that almost none of the environmental treatments had produced the expected results. Since then, proposals for similar treatments in water bodies for various purposes, including control of alien aquatic plants, continue to be made, even though the products and techniques employed would not appear to have changed. However, in that no recent assessments on these treatments have been run and due to the uncertainties concerning their effectiveness, none of the recent proposals of which we are aware have received funding.

Figure 59

© A. Dutartre, Irstea

Sediment treatment in a water body by projecting chalk powder mixed with water.

■ Water salinity

Freshwater aquatic plants are fairly sensitive to water salinity and regress to the point of disappearing when salinity increases. In certain littoral wetlands, the layout of sites may make it possible to inject or reinject water with varying degrees of salinity, particularly along the Mediterranean coast in the Languedoc-Roussillon region or along the Atlantic coast in western France. This technique may be a means to reduce the expansion of submergent and amphibious plants in these environments.

Among invasive plants, water primrose tends to colonise a wide array of wetlands. Lab tests by Grillas *et al.* (1992) on the resistance of large-flower water primrose to salt revealed that biomass production was impacted starting at salinity levels greater than 2 grammes per litre. On the other hand, it would seem that creeping water primrose (*L. peploides*) can accept up to 10 g/L (Mesleard and Perennou, 1996). In analysing the effectiveness of salt as a means of regulating the species in the Camargue area, Dandelot *et al.* (2005) noted reductions in biomass of almost 50% between the control plots and the treated areas on two of the three experimental sites (irrigation canal and pond), indicating a negative effect of salt on growth. However, the results on the third site (a marsh) were less clear and the authors concluded that "it was not possible to determine precisely the effectiveness of the technique", "even though the growth of the primrose beds was indeed slowed by the treatment. The effectiveness of the salt treatment was enhanced when combined with draining of the invaded site." Finally, the increase in salinity produced no perceptible effect on the invertebrate communities in the irrigation canal.

More recent work by Thouvenot *et al.* (2012) showed, under laboratory conditions and for the ends of plant stalks, that large-flower water primrose was sensitive to salt starting at 6 g/L. During this work, the same experiments carried out on parrot-feather watermilfoil revealed considerable differences in the resistance levels of the two species to salt. Increases in salinity (1.3 and 6 g/L) resulted in reductions in the photosynthesis and the growth of water primrose, but had no comparable effect on parrot-feather watermilfoil. The authors concluded that the reactions of the species depended on the season and on how each plant developed. Parrot-feather watermilfoil was deemed to be better suited for colonisation of brackish waters.

The potential for regulating water primrose using brackish water has been tested since 2013 in a section of the Brière marshes that in the past were naturally subject to tides and are currently heavily colonised by primrose. The three-year research programme comprises six, successive entries of brackish water, the first of which took place from the end of September to the beginning of October 2013. The monitoring programme intended to assess the effectiveness of the technique for water primrose and its impacts on water quality and fish communities (Thabot, 2013) effectively revealed plant mortality on certain sites. However, given the timing fairly late in the season, the mortality may have corresponded to the normal life cycle of water primrose, i.e. it could not be clearly attributed to the salt.

Following the first test, successive releases of brackish water were carried out starting in July 2014 in order to provoke a prolonged period (several weeks) with high salinity levels (10 to 20 g/L). This second test had a significant effect on the beds of water primrose with high levels of mortality. The monitoring also revealed high impacts on fish populations, including considerable mortalities on certain sites, probably due to the confined nature of the area receiving the brackish water from which the fish could not escape. The analysis of this full-scale experiment is still in progress and will not be complete until an assessment has been run on the spring regrowth of the water primrose in the tested areas and on the fish populations in the marshes. However, whatever the results of the analysis, this regulation technique for water primrose could be used exclusively in the sections of the Brière marshes closest to the Loire estuary and would have to be repeated each year in order to regulate primrose populations over the long term.

The effects of salt on groundsel bushes have been studied in Spain (Caño *et al.*, 2014). The species has a high tolerance level for saline environments, however its abundance is negatively correlated with the salinity level. High concentrations of salt in the environment are thought to have a moderate effect on plant mortality, but significantly reduce its growth rate and seed production. Greater loss of leaves was also observed under highly saline conditions. The combined effects may reduce the resistance of the species to pests such as fungus and scale insects. On the basis of these studies, comprehensive management techniques for ecosystems colonised by groundsel bushes, e.g. salt marshes, are currently being tested.

Recent work indicates that populations of Asian knotweed (*Fallopia* spp.) have been observed in coastal areas and salt marches in the United States (Richards *et al.*, 2008, quoted by Rouifed *et al.*, 2012). This capability of colonising saline habitats would seem to correspond to a tolerance on the part of the plants rather than to an adaptation to saline environments. For the thesis by Soraya Rouifed (2011), experiments were carried out to determine the degree to which Asian knotweed can tolerant salt stress.

In a first set of tests, adult plants from the three taxa (*Fallopia japonica, F. sachalinensis* and *F. x bohemica*) were subjected to treatments spanning a wide range of salt concentrations from 0 to 300 g/L, for a period of three weeks. In the second test, *F. x bohemica* plants were subjected to concentrations of 120 g/L following cutting of the aerial parts of the plants. The results of the tests showed that the aerial parts of Asian knotweed are sensitive to the highest concentrations, starting at 120 g/L, and that the biomass of their roots is significantly reduced by concentrations starting at 30 g/L. In addition, regeneration of treated *F. x bohemica* is delayed 20 days compared to the control group. “Saline shock” treatments, though somewhat effective under laboratory conditions, are not sufficient to prevent plant regeneration and the use of salt at concentrations exceeding 100 g/L would not appear feasible as a management technique for Asian knotweed in natural environments.

■ Light

Access to light, the indispensable factor for photosynthesis, determines the distribution of plants in aquatic environments, examples being light passing through water for submergent plants and light filtered by trees along rivers. Shallow waters in lakes and rivers are biotopes with high potential for submergent and emergent plants.

Among both native and alien species, needs in terms of light vary significantly. Some alien species need great amounts of light, for example water hyacinth and water primrose, whose beds are generally very sparse when riparian vegetation along rivers is dense. Other species can grow with less light, e.g. curly waterweed and large-flowered waterweed, which means they can colonise areas greater than five metres deep in lakes with highly transparent water.

It is possible to reduce or eliminate the available light for the targeted plants using different techniques such as tarps, subaquatic screens, dyes or management of riparian vegetation. A further technique consists of introducing burrowing fish, e.g. carp, a solution occasionally proposed to limit the growth of hydrophytes in lakes. Burrowing by such fish results in fine sediment particles being suspended in the water, which significantly increases turbidity, however this technique can be used, even in the best of cases, only in lakes where the increased turbidity does not hinder other uses.

All of these techniques have important limitations to their use.

Tarping has been used frequently in the past in attempts to eliminate the development of certain monospecific communities of terrestrial invasive species such as Asian knotweed and amphibious species such as water primrose (see Figure 60). This technique, often tested under experimental conditions, was judged insufficient in many cases because plants grew back through the tarp (knotweed) or recolonised the site once the tarp was removed (water primrose). However, it is still used and can produce worthwhile results on the condition that the sites are subjected to regular and relatively time-consuming monitoring and upkeep. The maintenance work on the tarp must be accompanied by additional management efforts, such as planting of other species (see the work done by the COEUR association in the Côtes-d'Armor department to manage knotweed in the management project in volume 2, page 91). Another tarping technique was successfully tested on water primrose by the “green team” at the Vistre public river-basin territorial agency (see the management project in volume 2, page 67). The tarps were put in place for short periods (10 to 15 days) during the summer on land colonised by the primrose. In the Mediterranean climate, the black tarps greatly raised the temperature underneath, thus weakening the plants and facilitating their uprooting once the tarp had been removed.

Figure 60

Installation of tarps and willow cuttings in an effort to manage knotweed.

Subaquatic screens intended to prohibit the colonisation of lake bottoms by submergent plants are a technique developed in North America over the past 40 years. These "benthic barriers" are generally used in sections of lakes heavily used by humans (for mooring, fishing, swimming). The term is used for various types of screen and sheets, some waterproof, some not. An array of materials, including burlap, plastic sheets, perforated black sheets, Mylar, woven synthetics, etc., can be used as barriers (Dutartre and Jan, 2012). Other materials have also been mentioned for this purpose, e.g. the bottom sheets for ornamental basins and pools or felt-type materials[9]. These screens have extremely variable service lives and maintenance conditions, up to 15 years for some if they are regularly cleaned to avoid clogging, an example being the type of screen installed in one of the basins in the port of Sainte-Eulalie (Landes department) to eliminate the beds of large-flowered waterweed (Dutartre and Jan, 2012). Some are biodegradable. Five hectares of burlap (a biodegradable geotextile) were placed in a section of Lough Corrib in Ireland as part of an effort to manage curly waterweed (see the management project in volume 2, page 27, and Figure 61). Very little information is currently available on the effectiveness of barriers over the mid-term. Given their cost and the anthropogenised nature of the resulting sites, they would seem to be reserved for areas heavily used by humans, such as ports, boating sites and areas in the immediate vicinity. By blocking the colonisation of these sites by submergent plants, they can contribute to reducing the "supply" of cuttings of plant stalks created by boat propellers to other areas of lakes.

Figure 61

Installing a biodegradable geotextile in Lough Corrib (Ireland).

9. *http://www.ecy.wa.gov/programs/wq/plants/management/aqua023.html*

The use of water dyes is another technique proposed by various companies. This technique was initially developed to reduce the growth of phytoplankton in ornamental basins and subsequently adapted to natural environments.

The products modify the colour of the water and thus limit the penetration of light in order to reduce or stop photosynthesis by the submergent plants. Their use is limited to small, stagnant environments and for aesthetic purposes. They must also be used prior to the development of the plants. This technique cannot precisely target any given species and must be repeated regularly. Its effectiveness varies significantly, depending on needs of the local species for light (see the management project in volume 2, page 50, and Figure 62).

Figure 62

A lake in the Netherlands receiving a dose of Dyofix.

A natural limiting factor for the development of macrophytes in watercourses and ditches is the shade created by riparian vegetation along the banks (Dawson and Kern-Hansen, 1979). Research in this field indicates that management of riparian vegetation could be an effective means to limit the communities of submergent macrophytes in these environments. For rivers less than 25 metres across, shade limiting 50% of the light would be sufficient to reduce plant development (both native and alien) to the point that they no longer create any significant hydraulic modifications. A sizeable percentage of invasions by knotweed and balsams along river banks is probably due to management methods for riparian vegetation over a number of decades that removed too many trees from the banks, thus greatly increasing the light reaching the soil and encouraging the establishment of opportunistic species.

A return to management techniques allowing the creation of denser riparian vegetation and consequently more shade could contribute to reducing the proliferation of certain plants. However, this would cause major problems for techniques currently used to maintain the edges of aquatic environments involving machines that would be severely inhibited by significant tree growth along banks.

■ Raising water levels

In some cases, it is possible to raise the water level over a long period, for example in reservoirs and in some lakes and ponds where the water level can be controlled by a dam. The rise in water level reduces the amount of light transiting the water and floods the banks, thus reducing the growth of submergent plants and riparian vegetation. For example, the work by Wallsten and Forsgren (1989) on Lake Tämnaren to the north-west of Stockholm showed that six years after a 30 to 50 cm increase in the water level, colonisation of the 35 square kilometre lake by plants had been sharply reduced. Calculations of surface areas using aerial photographs revealed reductions of over 80% for common reed (*Phragmites australis*) and the yellow water lily (*Nuphar lutea*), and over 60% for the common club-rush (*Scirpus lacustris*). The two submergent species in the lake, Eurasian watermilfoil (*Myriophyllum spicatum*), a native species, and Canadian waterweed (*Elodea canadensis*), an alien species, together occupied 236 ha of the lake in 1973, but were observed on only a small number of sites in 1983.

Similarly, research on the relation between the water level and the productivity of the main macrophytes in Lake Grand-Lieu (Marion *et al.*, 1998) revealed a clear correlation between the two factors, where higher water levels resulted in lower productivity. However, no clear correlation could be found between the water level and the surface area of the plant beds studied because the changes in surface area were in some cases due to other factors, e.g. eating of the plants by coypus.

In a guide drafted on water-primrose management in the Mediterranean region, Legrand (2002) wrote that a one-metre increase in the water level of a lake managed by the Fishing federation of the Hérault department eliminated the water primrose by exceeding its "flood tolerance". Whatever the case may be, this technique can be used exclusively in those cases where the dam controlling the water level can handle the greater mass of water, where the rise in water level does not risk flooding human activities and structures along the banks, and where the process is accepted by the lake owners and compatible with their management objectives.

■ Draining the water body

This is a technique traditionally used in drainable ponds for fish production. The objective is to facilitate harvesting of the fish, but also to reduce the accumulation of organic sediment due to its mineralisation in contact with the air during the time the pond is drained (see Figure 63). This technique may be used for all water bodies that can be drained, however caution is advised for water bodies containing invasive plants in order to avoid later problems in the water body itself and downstream.

In the water body, the type of sediment plays a critical role in the effectiveness of the draining technique. Even when there is no precipitation or input from the water table, the organic sediment often found in these environments can retain sufficient humidity over long periods to enable the survival of the subterranean parts (roots, base of stalks) of submergent and amphibious plants, meaning these plants can in many cases resume their growth once the water body has been refilled.

Figure 63

© A.Dutartre, Irstea

The drained pond in the town of Saint Pée-sur-Nivelle (Pyrénées-Atlantique department).

Among the invasive plants, submergent species such as the Hydrocharitaceae (tape grasses) resist poorly to drying and their foliated stalks, when exposed to air, are destroyed in a few days. The same is not true for amphibious species and for water primrose in particular. Its woody stalks resist drying much better and facilitate the regrowth of the species.

A few small ponds along rivers in western and south-west France were drained in the fall in an attempt to control the water primrose that had colonised the ponds. This technique, even when extended through the winter in the hope that the very low temperatures would kill the plants, did not produce the expected results. The water primrose was not totally eliminated from the ponds and, in at least one case, following a mild winter and a wet spring, the plants expanded their colonisation of the sediment before the pond was refilled, thus increasing their surface area in the pond.

The weather conditions during the time the water body is drained play an important role, e.g. freezing temperatures can be effective if the sediment is frozen to a sufficient depth. In the littoral marshes of the Languedoc-Roussillon region that have been colonised for a number of years by water primrose, more or less long drained periods, ranging from a few weeks to six months, have succeeded in reducing the invaded surface areas (Grillas *et al.*, 2001). The Mediterranean climate, with high temperatures and long periods without any precipitation, clearly contributed to the positive results.

In addition, the seed banks in the sediment can react quickly following the draining, thus enabling the development of species that are, in their majority, native and adapted to the new ecological conditions, but that then disappear rapidly when the water body is refilled. Among invasive plants, water-primrose seeds are capable of sprouting directly from the fruit lying on the sediment surface. Tests carried out in the lab and *in situ* have shown that initially the development of seedlings (and consequently of viable plants) was greater in organic sediment saturated but not covered with water (Dutartre and Petelczyc, 2005). Spring drops in water levels that can occur naturally or be caused by water management can thus encourage the development of water-primrose seedlings and of adult plants.

When draining a water body, particular attention must be paid to the diaspores of any invasive plants on site, e.g. entire plants, stalk fragments, the fruit and seeds of water primrose, in order to eliminate or at least sharply reduce diaspore flows downstream. Use of a filter (e.g. a fine screen) at the output of the water body during draining can reduce flows on the condition that the filter be regularly cleaned to avoid clogging (see Figure 64). Complete plants (with the possible exception of small floating plants such as duckweed and water fern) and the stalk fragments of submergent and amphibious plants can be fairly easily picked up by filters, however the probability that water-primrose fruit, that can float for several days up to a few weeks, and particularly the seeds will not be collected represents a major drawback to the filtering technique.

Figure 64

© A.Dutartre, Irstea

Filter to collect water-primrose fragments following uprooting in the Boudigau River, in the Landes department in 2002.

It should be noted that draining operations are subject to strict regulations. Draining of a water body created by a dam greater than 10 metres high or covering a surface area greater than 0.1 hectare requires an authorisation and may be carried out only during approved periods, generally in the spring and the fall. If the drained water flows directly or via a ditch or outlet to a category-1 river for fish, the water body may not be drained during the months of December to March included (decree dated 27 August 1999, modified by decree dated 26 July 2006). In addition, the Prefect may prohibit draining during water shortages. It is advised to check with the local authorities as to whether and when draining is authorised.

■ Cleaning and dredging

The accumulation of organic sediment in stagnant aquatic environments and those with a low discharge is a continuous process that can progressively modify the ecological functioning of these environments and hinder their use. The most common physical modifications are reductions in the depth of lakes and in the wetted cross section of rivers and ditches. This accumulation of organic matter proceeds in parallel with an increase in dissolved nutrients (notably phosphates and ammonia) that are consumed by the rooted plants, a positive factor for their development if the amount of light reaching the bottom is sufficient for photosynthesis.

Cleaning is part of the regular maintenance work on ditch networks in wetlands because it ensures the flow of water through the network (see Figure 65). The use of excavators with buckets up to two-metres wide is a means to combine cleaning operations with the removal of plant rhizomes, young plants, cuttings and, in some cases, the seeds stored in the sediment (Haury *et al.*, 2010). Particular attention should be paid to the timing of operations, taking into account local constraints, but wherever possible prior to the full development of the plants, particularly water primrose, in order to limit the dispersal of seeds in the environment. Similarly, precautions should be taken to recover the stalk fragments of plants that remain after cleaning.

The sediment extracted by excavators equipped with buckets is generally spread along each bank of ditches. The aquatic plants in the cleaned sections are extracted with the sediment. Spreading on the banks is not a problem for submergent plants because they rapidly dry out and die. However, amphibious plants, that are generally more resistant to drying, may be able to survive and even pursue their growth if the banks are fairly damp. The spreading of sediment can have other impacts, e.g. colonisation by groundsel bushes. On the slightly upraised and bare mounds, groundsel seeds carried by the wind can sprout. In regions where groundsel bushes are present, precautions should be taken to limit their expansion, for example by removing the extracted sediment from the site or by spreading it over a larger area when removal is not possible, or even by sowing the sediment (Damien, personal pub.). In all cases, it is advised to monitor the site following the work.

Figure 65

© A. Dutartre, Irstea

Cleaning a ditch.

To limit the risks of dispersal, parrot-feather watermilfoil and water primrose require specific treatment. Water primrose is particularly resistant. Sediment spread a few years ago without taking any special precautions on various sites, e.g. the Barthes area along the Adour River, is probably responsible for the later colonisation of nearby wet meadows. It was to avoid these problems that the Sèvre-Niortaise basin interdepartmental institution (IIBSN) published a brochure on what not to do (http://www.sevre-niortaise.fr/IIBSN_/wp-content/uploads/plaquettejussieenZH-2013.pdf). The plant matter of the concerned species extracted from ditches should be deposited outside of wetland areas. Finally, concerning water primrose, the fruit and seeds will be present in the spread sediment. Special monitoring of any sprouting and development of seedlings should be planned in order to launch, if needed, an intervention to remove the plants.

Cleaning and dredging are carried out in both natural and man-made water bodies in which the accumulation of sediment creates problems for the intended use (see Figure 66 a and b). The objective is generally to increase the depth and avoid problems by “cleaning up” the environment.

Figure 66

a, b © A. Dutartre, Irstea

(a) Dredging the Moysant Pond (Landes department).
(b) Cleaning the Forges reservoir at Ychoux (Landes department).

The main problem with these interventions concerns the extracted sediment. The volume of watery sediment can be considerable and spreading should take place only on sites correctly prepared for the temporary or definitive storage of the sediment. Sediment-extraction techniques may use an array of equipment depending on the type of environment, the type of sediment and the size of the area to be treated. Frequently employed devices include buckets, similar to those used on excavators, installed on pontoons and barges, or pumps, in some cases equipped with a cutter.

The presence of aquatic plants in the sectors to be dredged is a problem that must be taken into account when planning the intervention. In submergent, rhizomatous plants, e.g. Nymphaeaceae (water lilies), or alien, amphibious plants (parrot-feather watermilfoil, water primrose), the solidity of the tangled rhizomes/stalks can slow the work by hindering extraction (the mix of plants and sediment can block the bucket or the cutter). In that dredging can severely modify the ecological conditions in biotopes, interventions can be followed by rapid reactions of seed banks. Concerning invasive plants, the only species producing seeds that can react to this type of ecological modifications are the water primroses and, to our knowledge, no observations on this point have been published.

That being said, the presence of invasive plants in the treated area means precautions should be taken during sediment extraction and disposal, and the water body and disposal site should be monitored. The techniques used to extract sediment can result in plant fragments that can potentially recolonise the area. Dredging should therefore be followed by manual collection of the plant fragments and regular monitoring over a fairly long period in order to intervene if necessary on any new growth (see the management project in volume 2, page 63).

On the disposal sites, draining of organic sediment can take some time, meaning that plants may grow in the storage basins. Monitoring is required in order to intervene if necessary. For example, monitoring of the disposal sites for sediment extracted from a section of the Sèvre-Niortaise River, where water primrose has been managed for over ten years and consequently no large beds still exist, revealed in 2014 strong growth of water primrose on sediment that had been dredged (the plants were not dredged) a few months earlier in March (see Figure 67). This colonisation is an additional problem requiring management on the disposal sites.

Figure 67

© Nicolas Pipet, IIBSN

Water primrose developing on dredged sediment.

On the other hand, monitoring of the disposal sites for sediment recently extracted from the Marans canal that was severely infested by large-flowered waterweed did not reveal any new growth of the species (see the management project in volume 2, page 15). The water primrose noted in the sediment extracted by hydraulic dredging was rapidly eliminated by replanting the sediment (Fonteny, personal pub.).

■ Increasing current velocities

Local systems designed to direct or accelerate water currents have been used for decades to reduce the erosion of certain sections of river banks or to guide the flow of water toward the centre of the river bed. They can also be used to modify the local flow conditions to the point of limiting certain plant colonies, depending on the type of sediment and how solidly the plants are rooted in the sediment. However, these systems are difficult to create and to maintain, and their impacts on plants are extremely local (see Box 24).

The links between sediment types, current velocities, aquatic-plant morphologies and root systems are fairly well known, which means it is possible, at least approximatively, to foresee which biotopes will be colonised by which species. Certain species clearly prefer environments with fast currents (many Ranunculus species, e.g. water crowfoot), others tend to colonise organic substrates and are generally found in stagnant environments or those having very slow currents.

Most invasive aquatic plants fall into the second group (Peltre *et al.*, 2002). Amphibious species such as water primrose and parrot-feather watermilfoil, and submergent species such as the Hydrocharitaceae (tape grasses) almost invariably colonise biotopes that are stagnant or have very slight currents.

The idea of rapidly increasing river discharges (“artificial flooding”) is also occasionally mentioned as a means to control aquatic plants.

Once again, the links between the hydrological regimes of rivers and the development of aquatic plants are now fairly well understood. Major floods destabilise sediment and can pull out the parts of the plants (stalks and root systems) in the biotopes subjected to the flows. For example, the development of macrophytes was monitored in the Charente River (Dutartre *et al.*, 1994). No major winter floods occurred in 1992, but the two following years saw strong floods and plant colonies in the river were sharply reduced. At the Nersac monitoring point downstream of Angoulême, the percentage of colonised spots fell from 70% in 1992 to approximately 35% in 1993 and less than 20% in 1994.

River flows and the adaptation capacities of certain aquatic-plant IASs

The links between the hydrological regimes of rivers and the development of aquatic plants are now well understood. However, no absolute rules may be derived from this information. For example, water primrose is capable of surviving in certain shallow, river biotopes where continuous flow rates can reach 30 to 40 centimetres per second. However, under these conditions, plant characteristics are somewhat different. The work by Charbonnier (1999) on water-primrose development dynamics in different biotopes showed that plants established on a primarily sandy substrate and subject to currents at a monitoring point in the Isle River, a tributary of the Dordogne River, had significantly lower productivity rates, biomass, stalk sizes and number of branches than plants in stagnant environments also studied. Average stalk lengths did not exceed 50 cm, whereas they reached two to four metres in other environments.

The Hydrocharitaceae species are also capable of resisting, temporarily and permanently, flow conditions that would theoretically appear to be incompatible with relative weakness of their stalks and root systems. For example, large-flowered waterweed (Egeria) may be observed in the Dordogne River, downstream of Bergerac, where currents exceed 50 cm per second (Breugnot, 2007). The beds are generally located just downstream of boulders or outcrops creating biotopes without any current where the plants can put down roots in the fine sediment deposited there. The same is true for curly waterweed that has been observed in the Adour River, just downstream of large beds of Ranunculus species where it is protected from the current (Delattre and Rebillard, 1996).

Under such conditions, neither water primrose nor the Hydrocharitaceae species can produce sufficient amounts of biomass to become pests, but these colonies contribute to the flow of propagules travelling downstream.

The floods at the end of the spring in 1994, even though they were not as strong as the winter floods, had a major impact on submergent plants just starting to grow. The evaluations of plant biomass at the Nersac monitoring point, taking into account the distribution and biomass data of the various species, revealed a drop in biomass by a factor of ten between 1992 and 1993 (approximately 1 000 tons of dry matter in 1992, 105 tons in 1993).

Monitoring of colonisation by large-flowered waterweed since 2010 by the Thouet valley board also showed the high impact of winter floods on the development of hydrophytes, including Egeria, in the river (see Figure 68). Starting with the strong floods in December 2011, followed by other significant floods in 2012 and 2013, plant colonies decreased greatly in size to the point of disappearing completely from certain monitoring points (Charruaud, personal pub.).

Figure 68

© A.Dutartre, Irstea

Stalks of large-flowered waterweed (Egeria) uprooted by floods (Thouet River, Deux-Sèvres department).

This type of change in hydrological regimes can be considered a natural means to control certain plant colonies, both native and alien, by eliminating the least rooted and the most fragile species. That is why, theoretically speaking at least, artificial floods created at the correct moment, e.g. when submergent plants develop their stalks and become easier to uproot, could produce the same results. Unfortunately, in the current context of efforts to improve the quantitative management of water, this approach is virtually inapplicable. It should also be noted that aquatic plants uprooted by floods can have serious consequences on human facilities. In addition to masses of plants gathering at certain weirs, dams and locks, where they can hinder flows, water intakes at facilities can be blocked, with variable consequences. Water intakes at nuclear power plants can occur, leading to the temporary shutdown of reactors. This type of incident occurred at the Cruas nuclear power plant on the Rhône River in December 2009 (Carrel, 2009), when 50 cubic metres of plant matter blocked the intakes, resulting in an level-2 incident (on a scale of eight levels from 0 to 7, with levels 1, 2 and 3 representing incidents and levels 4 to 7 representing accidents). Virtually all the biomass consisted of Nuttall's pondweed (*Elodea nuttallii*). Among the half-dozen other hydrophytes noted was large-flowered waterweed (*Egeria densa*). These two Hydrocharitaceae are among the species that have difficulty resisting currents.

Interventions to limit plant growth

The types of curative intervention available to managers are now fairly well known, but they have a number of limits and produce impacts that must be understood if we are to improve these management techniques. A number of books are available on the subject, but of particular interest is the management manual for invasive alien plants in aquatic environments in the Loire-Bretagne basin (Haury *et al.*, 2010).

■ Manual interventions

Manual interventions have existed since it became necessary to launch control operations against aquatic plants, probably around the beginning of the 1900s. Changes in the cost of labour, in the available techniques and notably the use of herbicides since WWII, as well as the rather negative perceptions of manual labour (tiring and often dirty) have resulted in the progressive halt to manual interventions on many sites.

In a report on mowing plants in rivers, Isambert (1989) discussed the various techniques used in the Seine-Normandie basin to control aquatic plants. Manual maintenance work was carried out on some 15 rivers with a cumulative length of approximately 250 kilometres. Virtually all the work was organised by river boards. The tool used for this work was a “châtelaine”, a cutting bar drawn manually across the river bed by two workers on the river banks. This type of tool is suitable for rivers less than 12 metres wide and where the banks are easily accessible. Two experienced workers could mow approximately one kilometre of river per day. The use of part-time labour and volunteers was also mentioned. Unfortunately, no information is available on recent updates to these traditional techniques.

Concerning alien plants, the management plan for aquatic plants in the lakes and ponds of the Landes department, prepared in 1989 for Géolandes (Dutartre *et al.*, 1989), recommended manual uprooting for “sections invaded to a limited degree” by curly waterweed, water primrose and parrot-feather watermilfoil.

At that time, the proposal for manual interventions elicited a number of negative reactions from both elected officials and the concerned technical services. Some people perceived the proposal as a “return to the Dark Ages” given that an array of available machines and herbicides were seen as effective in meeting the management needs of the plants. Other, rather excessive, reactions spoke of prison camps and the use of inmates to do the work.

A number of practical demonstrations and experimental interventions were organised on several sites to dispel those impressions. For example, the manual interventions in 1992 and 1993 on the banks of the Noir Pond (a nature park) and another nearby pond produced five cubic metres of water primrose the first year, but just 0.05 cubic metres the second, a reduction by a factor of 100 that demonstrated the effectiveness of the work (see Figure 69).

Figure 69

a, b © A.Dutartre, Irstea

*Manual uprooting of large-flower water primrose (*Ludwigia grandiflora*) by volunteers in the Noir Pond (Landes department).*

At the same time, efforts to explain the rationale, noting for example the risks of cuttings being dispersed by the machines used and the lack of selectivity by herbicides, resulting in the disappearance of not only the targeted plants, but of all plants on the site, produced a change in opinions concerning manual interventions. This change was fairly rapid because in just 20 years, manual interventions were regularly organised for amphibious plants, water primrose on a large number of sites and parrot-feather watermilfoil on a smaller number sites, on lakes in the Landes department, the Marais Poitevin marshes, the Brière marshes and many other sites, primarily in western France.

An analysis of the change revealed the main reasons (Menozzi and Dutartre, 2007), namely the precision and effectiveness of manual work, leaving the “non targeted” plants in place, something that mechanised techniques cannot achieve. The authors even wrote about “manual uprooting of water primrose, an age-old technique that is truly an innovation” (Menozzi and Dutartre, 2008). It is, however, clear that this type of intervention is justified only under certain conditions, e.g. at the beginning of a colonisation, for collection of plant fragments remaining on site after a mechanised intervention or in areas that machines have difficulty in reaching, and on the further condition that the quantities of biomass to be extracted are not excessive.

The physical difficulties involved in manual interventions should not be exaggerated. The work is physically demanding, occasionally under difficult outdoor conditions, but a considerable number of the negative opinions mentioned above concerned poorly organised work sites involving persons having received little or no training. Similar to any other type of intervention, the objective is to correctly organise the work and consequently to reduce its difficulty and any harsh conditions.

The easing of the physical conditions may concern, for example, the movement of personnel if the sediment is highly liquid or the water is excessively deep. Boats may be necessary, if possible with flat bottoms to improve stability. Similarly, transportation of the extracted plants from the site may be a complex and tiring task. Suitable bags, sufficiently large boats and equipment on the banks to discharge the plants from the boats are examples of means that can be employed to facilitate the task. Considerable efforts have been made by IIBSN to improve the operational conditions for the team working on water primrose in the Marais Poitevin marshes. A boat was purchased to provide the workers with a place to eat out of the weather and to store their tools, etc. (see Figure 70).

Figure 70

© Nicolas Pipet, IIBSN

The work conditions for manual interventions were improved by providing the workers with a boat for their meals and to store their tools.

Other efforts to improve hygiene and safety include training sessions on first-aid and ergonomics, vaccinations and a safety manager to implement a Special plan for work safety and health (PPSPS) (Pipet and Dutartre, 2014) (see the management projects in volume 2, pages 34 and 67).

As already noted, these interventions concern almost exclusively amphibious species. Manual uprooting of submergent plants is possible, but more complex and less effective due to the fragile stalks and the deep water conditions in some cases. However, when manual interventions are carried out rigorously and taking the necessary precautions, in particular when a maximum quantity of plants and stalk fragments, even very small pieces, are removed from the site, the work is very effective and sharply reduces regrowth of the targeted species. Because of their targeted nature and the very limited use of equipment, their impact on environments and habitats is very slight and even inexistent because the non-targeted species are not affected, which means they can continue to develop because the targeted alien species no longer exerts any competitive pressure within the living community. For example, the regular management work on water primrose in the Marais Poitevin marshes paved the way for the re-installation of various native submergent species or those having floating leaves (Pipet, personal pub.).

On banks where invasive plants have appeared, they can be uprooted by hand as long as they have not developed too much, otherwise tools (spade, pickaxe, hoe) must be used. It is very important to remove the entire root system, particularly for suckering plants, to avoid regrowth and even multiplication of the plants if removal is only partial. The uprooted plants must be completely removed from the site to avoid leaving any plants capable of putting down new roots. Manual cutting of these species is also possible if the installation process has just begun or if the site is only sparsely colonised, or in those cases where, given the ecological value of the habitats or of the native species, mechanised interventions are not possible (see the management project in volume 2, page 99). Depending on the diameter of the stalks, the plants can be cut using billhooks, sickles, secateurs, saws, chainsaws, etc. (see Figure 71).

Figure 71

© ONCFS

Manual cutting of groundsel bushes using a billhook in the Lège-Cap-Ferret National salt-meadow nature reserve.

This work can be done by volunteers under management, temporary or permanent staff of local governments, "make-work" companies for the unemployed or private companies. Whatever the case may be, minimum training is required concerning identification of plant species, the precautions required during plant uprooting and transport, and safety conditions for work in areas that are generally difficult to access. The existence of technical specifications for each type of intervention facilitates the work and improves its effectiveness. For a number of years, several private firms have specialised in manual techniques and offer their services to local governments.

In a number of other cases, very small firms offer manual uprooting by scuba divers. Generally speaking, the treated sites are fairly limited in size, e.g. ports where the presence of docks, chains and other factors hinder or make impossible the use of machines to harvest the plants. This technique can be used in deep waters, however its cost even greater than other manual techniques, the specialisation of the personnel employed and the difficulties involved in subaquatic work mean that this technique is restricted to limited operations on sites having high added value.

■ Mechanised interventions

Mechanised techniques to manage submergent plants have been used since the 1920s. The equipment used has very often been adapted from agricultural devices (cutting bars used for mowers, conveyor belts, etc., see Figure 72). The wide range of equipment can be used in many types of situation (Dutartre and Tréméa, 1990). The “châtelaine” mentioned above for manual interventions (Isambert, 1989) is still used today for certain types of work. The weighted cutting bar or bars can be dragged along the bottom by a boat.

Figure 72

© A.Dutartre, Irstea

Cutting bar that is dragged along the bottom by a boat.

Some of the devices used simply cut (mow) the plants. They generally consist of an upside-down T-shaped cutting bar positioned at the front of the boat. The horizontal bar at the bottom enables the boat to cut its way through the plant beds (see Figure 73). On this type of device, the horizontal cutting bar can rarely cut more than one metre below the water surface. Until recently, the cut plants were not harvested. They simply floated along with the current or the wind, either downstream in rivers, usually ending up at dams, or to the edges of lakes, not far from where they were cut. Given that one of the main criticisms of mowing was the fact that masses of cut plants were left in the environment, at the risk of causing oxygen deficits when they rotted or proliferating via cuttings, a number of manufacturers now propose harvesting systems that can be set up on the boats once the cutting bars have been dismantled (see Figure 74). Other devices specifically designed to harvest the plants are now available and used by a few specialised companies in addition to the mowing boats for large operations in lakes.

Figure 73

© A.Dutartre, Irstea

Mowing boats equipped with cutting bars in front and on the side.

Figure 74

© A.Dutartre, Irstea

Harvester rake for cut plants.

The most recently developed systems can simultaneously mow and harvest the plants. Given their size and the difficulty of moving them, these combined harvesters are better suited to stagnant environments or those with slight currents, having regular bottoms (Dutartre and Tréméa, 1990). In addition to U-shaped cutting bars (two vertical and one horizontal bar, or simply one vertical bar), these systems are equipped with a least one conveyor belt to extract the plants as they are cut. These systems are capable of cutting at depths of up to two metres (see Figure 75).

Figure 75

a © N. Pipet, IIBSN
b © A.Dutartre, Irstea

Harvesting vessels.

On the largest combined harvesters currently available (see Figure 76), two other conveyor belts can be used to store and then unload the plant mass. The plants can be unloaded either directly to the shore or to containers on barges near the harvesting zone, in order to reduce the time required for travel by the harvester. Smaller harvesters are equipped with only the one conveyor belt to extract the plants from the water. Storage on board and unloading of the plants are carried out by on-board personnel. Cutting bars are relatively fragile. They break

Figure 76

Harvester with the cutting bars and the conveyor belt used to extract the plants from the water.

easily when they encounter obstacles, either along the bottom or, for example, stakes and various other objects installed by humans. The areas in the aquatic environments to be harvested should be inspected to pinpoint where there is a risk of accidents and broken tools.

Depending on their size, these combined harvesters can store up to several cubic metres of plants and transport them to the unloading site. The fact that they can simultaneously cut and harvest is an advantage compared to systems limited to just one function, particularly for the management of submergent plants whose cuttings are likely to cause subsequent proliferation of the plants. The combined systems limit the production of stalk fragments and the conveyor belts are generally effective in limiting the release of cuttings to the water.

Mowing and harvesting work is most often carried out on sites where human activities (navigation, fishing, hunting, etc.) are hindered by dense beds of native and/or alien hydrophytes close to the water surface. The work is of value in that it facilitates human activities, but its effectiveness over time is highly variable depending on the site and the species. At best, it can be effective for up to one year, but in general it does not exceed a few months, the time required by the plants to regrow and reach the water surface.

The risks involved and the side effects of mechanised interventions are well known. In particular, it is not possible to select among the plants cut and harvested. In addition, the passage of the harvesters momentarily stirs up the upper, fluid layer of sediment. Finally, the invertebrates living among the extracted plants are also removed from the environment, as well as larger vertebrates such as turtles and fish that may be trapped in the plants.

To determine the damage caused to fish populations by harvesting operations, a study was conducted under the responsibility of IIBSN in 2002 and 2003 on Lake Noron, along the Sèvre Niortaise River downstream of Niort (Dutartre *et al.*, 2005). This lake, used essentially as a tourist attraction, was heavily colonised by mostly native hydrophytes, including the largely dominant rigid hornwort (*Ceratophyllum demersum*). A preliminary review of the literature indicated that a majority of the fish captured during this type of intervention were juveniles (born the year in question) and that losses in terms of numbers or biomass varied from 2% to 25%, depending on the author.

The study confirmed the data on the age of fish and the losses calculated over the reach in which Lake Noron is located amounted to 5.6% and 1.3% respectively for the years 2002 and 2003, which corresponded to the “low estimate” indicated in the literature. These relatively low values would seem to indicate that for the reach in question, the regular harvesting work undertaken for tourism activities in Lake Noron had a very low, if not negligible impact on fish populations. Though the observations made over the two years of experimentation are not sufficient to draw any firm conclusions, it was nonetheless noted that a smaller quantity of fish (in number and biomass) was captured during afternoons and when the harvester travelled from upstream to downstream. This information suggests possible modifications in harvesting techniques in this type of environment. However, depending on the period when the work is done and the type of environment, capture rates can increase and it is important to pursue this type of observation on the side effects of this type of intervention.

Regular harvesting is conducted in lakes that are heavily used for tourism. For example, that is the case of Blanc Pond in the southern section of the Landes department (see the management project in volume 2, page 23), where the work makes possible the continued use of the lake for summer tourism, fishing and hunting by clearing each year a part of the very dense beds of curly waterweed that have colonised over 100 of the 180 hectares of lake. Noting a reduction in the extracted quantities of biomass over the past few years, the Géolandes management board funded a study to assess the impacts of regular harvesting and determine the reasons for the apparent slowing in the development of the species (Bertrin *et al.*, 2014). The investigation did not detect any notable differences in water and sediment quality between the different monitoring points (colonised and non-colonised areas, harvested and non-harvested areas, etc.), which could have explained the trend and contributed to modifications in the management strategy for the species in the lake.

Other devices such as mower buckets, claws, etc., installed on a hydraulic arm attached to a land vehicle (tractor, excavator) or a floating vehicle (boat, barge, etc.) can be used to uproot and/or remove submergent and amphibious plants. Buckets used for this purpose are often equipped with screens to let water and fine sediment fall back into the water during the extraction of plants and they also have more or less evenly spaced teeth to better grasp the plants (see Figures 77, 78 and 79).

Figure 77

© Nicolas Pipet, IIBSN

A bucket designed for the mechanised uprooting of water primrose.

Figure 78

© A.Dutartre, Irstea

A claw used in the Marais Poitevin marshes to transfer water primrose from a barge to a trailer.

Figure 79

© A.Dutartre, Irstea

Uprooting of water primrose using a claw in Blanc Pond (Landes department).

This type of equipment is capable of extracting great quantities of plant biomass and conveying it directly to trucks for transportation to the disposal site. The work would appear to be more effective when the system is installed on a floating platform rather than on land (Haury *et al.*, 2010). Buckets and claws can be single or two-sided and the space between the teeth depends on the type of plants extracted (teeth slightly separated for submergent plants, more distant for amphibious plants). Equipment configuration and operator dexterity all contribute to the effectiveness of interventions.

This technique can be used to remove all or part of the root systems of the plants, but it also pulls up variable quantities of sediment around the roots, thus creating temporary pollution that depends largely on the type of sediment. Consequently, the type of sediment, which can range from muds with high levels of organic matter to mineral elements having highly variable grain sizes, is a factor that must be taken into account when attempting to assess the potential impacts, the effectiveness of the intervention and how to recycle the extracted matter.

The risks of creating cuttings due to fragmentation of plant stalks during this type of intervention is fairly high (Haury *et al.*, 2010) and should be taken into account during the assessment of the potential impacts, namely the dispersal of plants from the work site.

Native plants along banks may be cut or mowed using the equipment currently available for maintaining road sides and river banks (see Figure 80). However, the same techniques should be used very sparingly for most invasive alien species, e.g. groundsel bushes (*Baccharis halimifolia*), giant hogweed (*Heracleum mantegazzianum*), garden balsam (*Impatiens* spp.) and the Asian knotweeds (*Fallopia* spp.), because they risk stimulating the plants. Repeated cutting over several years, on the other hand, can exhaust these species, notably groundsel bushes and knotweed, and eliminate any seed banks (Haury *et al.*, 2010). Particular care must be taken when transporting cut stalks to avoid losing them en route and thus reduce subsequent dispersal of the alien species.

Figure 80

© A.Dutartre, Irstea

Flail cutting of a river bank in the town of Dropt (Gironde department).

The use of mulching machines is possible on fairly flat terrain that is sufficiently stable to support the passage of the large, often tracked machines (see the management project in volume 2, page 102 and Figure 80). The shredded organic material remains where it was cut. This technique can generally not be used on river banks.

Using an excavator to uproot plants on banks can be very effective if it succeeds in pulling out the entire root system, which is fairly easy for species having superficial root systems, e.g. garden balsam, but much harder for species with deeper root systems such as knotweed. This technique should therefore be limited to interventions on fairly small sites where important ecological issues are involved (Haury *et al.*, 2010). Any soil remaining from grading operations can be removed taking care not to allow any stalks or rhizome fragments to escape during transportation. Concerning knotweed, care must be taken if the excess soil is to be reused elsewhere given the capacity of rhizomes to produce new plants. Soil drawn from sites colonised by knotweed, transported to other sites and used as land fill without paying attention to the rhizomes is one of the reasons for the rapid dispersal of knotweed species in continental France.

Management of soil contaminated with rhizome fragments is very difficult. Prohibiting any future use would not seem feasible because it would then be necessary to store the soil somewhere, for example by burying it. Currently, no generally applicable solutions are available. Depending on the type of soil, it would be possible to screen it to remove the rhizome fragments. That is a solution proposed in the U.K. to manage knotweed-contaminated soil (see http://www.wiseknotweed.com/japanese-knotweed-removal-treatment/screening-sifting/).

Experiments involving grinding the soil containing rhizomes, then burying it and covering the site with tarps until the rhizomes have completely decomposed have shown that it is possible to avoid the risks of dispersal (see the management project in volume 2, page 81). Though expensive, if conducted rigorously over fairly small sites, this technique is effective, however its use along rivers is problematic due to flooding and erosion risks, particularly for the plastic tarp that is an essential element in the success of the intervention. This technique cannot be used over long distances for cost reasons, however it is useful for installation sites upstream in river basins in order to prevent later colonisation downstream. It should be noted that a second mechanised intervention is required one or two years later.

Figure 81

© SMAGE des Gardons

Mechanised management of knotweed in the Gard department, a) soil grinding, b) grinding rotor in the bucket.

■ Information on the use of herbicides

For decades, herbicides were commonly used in France to control the growth of aquatic plants until they were totally prohibited at the end of 2009. The ban was preceded by a fairly rapid drop over a few years in the number of marketed products authorised for this specific use. The use in “aquatic environments” was made possible by a waiver concerning the general ban on herbicide use in or near water. Numerous debates and disputes arose around this use, particularly concerning the toxicity of these products (acute toxicity, persistence) for the living communities not targeted by the products and the contribution of this management technique for aquatic plants caused by plant-protection products used in agriculture to water pollution and consequently the drop in water quality.

Over the years, regulations were modified in an attempt to better control the environmental consequences resulting from the use of these agricultural inputs. A number of European directives contributed to these improvements, notably the directive EEC 80-778 on the quality of drinking water, which set maximum contamination levels, directive EEC 91/414 on marketing authorisation for plant-protection products, reinforcing the toxicological and ecotoxicological criteria for approval of new substances and programming the re-evaluation of older substances, and more recently the Water framework directive (2000/60/EC). The latter directive, adopted in the year 2000, requires that Member States achieve by 2015 good chemical and ecological status of surface waters and good chemical status of groundwater. Finally, the framework directive 128/EC (21 October 2009) established a framework for EU action in view of achieving sustainable use of pesticides. In France, the directive resulted in the launch of the Ecophyto plan developed during the Grenelle environmental meetings in 2008. The objective of the plan is to progressively reduce the use of plant-protection products in both agricultural and non-agricultural areas (see the pages concerning the plan at http://agriculture.gouv.fr/ and http://www.ecophytozna-pro.fr/).

The decree dated 12 September 2006 on the marketing and use of the plant-protection products listed in article L.253-1 of the Rural code rural stipulates the code of conduct governing the use of the products (http://www.ecophytozna-pro.fr/data/arrete_du_12_09_06_7.pdf). In particular, it makes mandatory pesticide-free zones at least five metres wide along or around all water resources (rivers, lakes, ditches and all permanent or seasonal resources shown as points, lines and dotted lines on the 1:25 000 scale maps published by the French National geographic institute (IGN). No pesticides may be applied directly in a pesticide-free zone. The width of the zone can vary from 5 to 100 metres, depending of the water resource and the type of product. The list of water resources protected by the decree may be modified by prefectoral order to take into account any special local conditions. The reasons for the order must be explicit (see for example the prefectoral orders for the Deux-Sèvres, Loire-Atlantique, Maine-et-Loire and Vendée departments, with explanatory documents that may be downloaded from http://www.sevre-nantaise.com/espace-publications/).

Management techniques using herbicides are still very common in many countries, including the United States and the U.K., for example. In the U.K., glyphosate is used to eliminate water primrose from the few sites where it has become established, even if *“repeated applications over many years are required to eliminate the plant. Tiny rhizome fragments can survive the treatment and form new plants that are easily overseen in the field”* (Renals, 2014).

It should be noted that these uses of herbicides generally do not attain the objective of eliminating the targeted plants often presented by the advocates of this technique and that their effectiveness is in fact limited to one or two years. In addition to their toxicity, these products are not selective, i.e. they completely clear the treated zone, thus making them a method that must be used “with the utmost caution” (Dutartre, 2002).

Necessary precautions

The objective of work on invasive plants should be to eliminate them (in the rare cases that is possible) or to control them (in the vast majority of cases) by taking the necessary precautions to make sure that interventions do not become an indirect cause of additional dispersal of the species. Among the potential problems inherent in many alien species, the capacity of small cuttings (fragments of stalks or rhizomes just a few centimetres long) to produce new plants is probably the issue to which environmental managers should pay the most attention.

This capacity has now been extensively assessed for hydrophytes such as the Hydrocharitaceae (tape grasses), amphibious species such as water primrose and species growing on banks along water such as the Asian knotweeds. That is why it is very important that the methods employed fragment the extracted plants as little as possible, or if fragments are produced, that additional measures be taken during the work to collect as many fragments as possible before they can disperse. Nets are often used to retain hydrophytic or amphibious species within the intervention zone (upstream and downstream in a river or ditch, the work perimeter in a lake) (Haury *et al.*, 2010), however the nets must be frequently cleaned. In certain special cases, for example in sections of a ditch, temporary cofferdams may be used to limit dispersal during the intervention.

In addition to the above techniques, it is now acknowledged that manual collection of any fragments remaining on site following the mechanised intervention is an indispensable step that improves both the quality of the work and its effectiveness over time. Collection of fragments by hand or using a dip net, called "skimming" in the book by Haury *et al.* (2010), is a way to gather fragments of all sizes and in places that are difficult to access. This technique is particularly effective for amphibious plants such as water primrose.

If knotweed is temporarily stored prior to transportation for its elimination (e.g. by burning), care must be taken to ensure that the plants do not touch the ground to avoid any risks of regrowth on the site. It is advised to lay tarps or to create a thick mat of branches from other species on which the knotweed can be laid to avoid any contact with the ground. Another possibility is a non-woven geotextile fabric that is not as heavy as a tarp and is not waterproof, thus allowing the cut plants to dry more rapidly (Reygrobellet, personal pub.). It is now proven that many knotweed stands can produce viable seeds, which means it is better to intervene before they flower (Haury *et al.*, 2010), thus reducing the dispersal of the species and possibly limiting the development of fertile hybrids (*Fallopia x bohemica*).

The site and equipment (machines, hand tools, equipment used by workers) must be cleaned at the end of an intervention to avoid the accidental dispersal of stalk fragments and of rhizomes. Particular care must be given to amphibious species and species growing on banks that can resist drying. Many introductions of water primrose and of Asian knotweed are caused by fragments transported by machines that were not cleaned (Haury *et al.*, 2010). Cleaning equipment (in particular high-pressure cleaners) should be a basic on-site component for companies or teams conducting interventions to ensure that cleaning effectively takes place before machines leave the site, thus limiting the risks of propagule dispersal.

Grouping of plants and their transportation away from the site to a temporary storage place before being definitively recycled also requires extensive precautions to reduce to a minimum the fragments left on site or escaping during the transport (see Figure 82). For example, a tarp can be laid on the bank where the barge lands to unload the extracted plants onto a trailer or truck. Any plants falling to the ground can be easily recovered and shipped following the cleaning of the site (see the management project in volume 2, page 70). Temporary storage on a bank, even on a tarp, involves serious risks of dispersal. The vehicles used for transportation must be selected using the same criteria, i.e. to avoid any dispersal of plants.

Finally, certain invasive plants produce seeds that represent a further means of dispersal, in addition to stalk fragments and rhizomes. Specific examples are water primrose among the amphibious species and groundsel bushes among those growing on banks. The size of seeds and the ease of dispersal by water and wind makes it virtually impossible to control them. The only solution is to intervene, where possible, prior to the development of the seeds (e.g. by cutting groundsel bushes before they flower) and to monitor sites from which invasive plants have been removed in order to react rapidly if new plants appear.

Figure 82

© Hélène Gervais, CEN Centre

Safe unloading of knotweed on a tarp.

Panorama of management techniques for animal species

The various methods, both direct and indirect, of controlling invasive alien animal species are listed in Table 9, page 182.

Direct control of animal populations

The main methods used to manage invasive alien animal species consist of limiting population numbers. An array of techniques are used, most of which are highly regulated and require authorisations.

■ Trapping

This technique is used to remove individuals and thus limit population numbers. Regulations governing the trapping of animals have undergone major modifications since the 1980s. Certain types of traps are totally prohibited (e.g. steel-jaw traps and firearm traps), others are regulated. The list of animals considered harmful and that may be trapped has been reviewed yearly since 2012 and is published in a ministerial decree. The types of traps are grouped in categories and those provoking immediate death of the animal must be approved. Trappers operating in the natural environment must have received an authorisation (granted following a mandatory training course), except for those persons trapping coypus and muskrats using cage traps.

The latter is the most commonly used type of trap for invasive rodents and American mink (see Figure 83). This category-1 trap is a non-lethal, selective device has limited impact on non-targeted, native species such as beaver, otter and the European polecat, etc. Trappers are required to check their traps daily. The conibear trap is used to kill invasive rodents. It is a category-2 trap and its use is prohibited in areas where beaver, otter and European mink are known to live. Consequently, it is rarely used in aquatic environments.

Figure 83

© Anaïs Borrell

An American mink captured in a cage trap.

The use of bow nets (see Figure 84) to capture invasive alien species of amphibians, reptiles, fish and invertebrates is also regulated. On public property, the use of bow nets is regulated on the departmental level (number and types of bow nets depending on the river fish category, recreational or professional use, etc.). In general, if the intention is to manage an invasive alien species, a prefectoral order is required.

The partial immersion of bow nets can limit their impact on native, air-breathing species because the animals can remain on the surface and not drown. Regular checks on bow nets are mandatory. However, partial immersion does not limit the impact on non-targeted water-breathing animals, e.g. fish or amphibian larvae. For this reason, in the framework of management work on red swamp crayfish in the Brière marshes, highly selective traps were designed precisely to limit the capture of sensitive species such as eels (Paillisson et al., 2013).

Figure 84

© Guillaume Koch

Partially immersed bow-net trap set to capture African clawed frogs.

A number of other selective traps have been developed or adapted for the management of invasive alien animal species. An example is the "Fresquet cage" designed to trap red-eared slider turtles (see the management project in volume 2, page 175). The trap is cage made of wire grid with, at the bottom, an entryway in the form of a tunnel (see Figure 85). Contrary to a bow net, it is placed on the bottom of the pond or lake to catch turtles that move and hunt along the bottom. The top of the cage always remains above the water surface to allow the captured animals to breathe (Cases, personal pub., 2014).

Figure 85

© SYMBO

"Fresquet cage" used to capture red-eared slider turtles.

Nets can also be used to catch fish as well as birds during their postnuptial moulting when they cannot fly. Their use in aquatic environments is governed by the applicable departmental regulations on fishing techniques other than angling. For birds, use of nets is subject to hunting regulations. In all cases, if the intention is to manage an invasive alien species, a prefectoral order is required. Great skill is required to net birds during the moulting period, however a large number of birds can be captured in a limited amount of time. This technique is difficult to use in urban settings and on sites where the general public is present in number, and it often results in misunderstandings if no prior effort was made to inform the public on the purpose of the management work.

Concerning the use of gillnets to capture invasive fish species, it should be noted that their effectiveness depends greatly on the targeted species. For example, they work poorly for the Wels catfish (Silurus glanis), due to its size and morphology. They also lack selectivity and are often a lethal technique for many species.

Finally, it is important to note that trapping is not an effective means to ensure control of invasive populations or to limit their impact on the environment. Further, side effects such as an increase in recruitment have been observed (the case for alien crayfish, Poulet, 2014).

Shooting

The elimination of IAS animals takes place during interventions conducted or managed by the responsible oversight agency (ONCFS, wolf-hunting officers, etc.). A shooting campaign requires a prefectoral order complying with article L411-3 of the Environmental code. Interventions to limit certain populations must take into account all safety measures and avoid impacts of the shooting campaign on other species. The most commonly used weapons are smooth-bore guns (12-gauge shotgun) and various calibre rifles (222 REM, 22 Long rifle, 17 HMR and 22 Hornet) (see Figure 86). These weapons may be equipped with scopes and moderators. Air rifles are also used to shoot American bullfrogs. it is now mandatory to use steel shot and not lead shot when shooting takes place in aquatic environments (ministerial circular dated 4 April 2006).

Figure 86

© JF. Maillard

Shooting campaign for ruddy ducks. Note that shooting on water or ice is prohibited due to ricochet risks. These interventions are strictly regulated and numerous precautions must be taken to avoid any risk of accident for the shooter and for the accompanying personnel.

Hunting and fishing

Hunting is a means to reduce the numbers of certain IAS animals. However, it is limited to the authorised hunting species, the annual hunting season and a valid hunting license is required. In 2014, six invasive alien vertebrates were listed as game and pest species, i.e. they could be hunted. The six were coypus, muskrats, American mink, northern raccoons, raccoon dogs and Canada goose. Because of potential confusion with European mink, a protected species, it is prohibited to shoot American mink in the eleven French departments where the species is present.

To the best of our knowledge, angling was never an effective means to manage fish, crustaceans or "frogs" designated as invasive and it could even worsen situations due to the dissemination of the captured animals. That being said, angling of non-native species of fish, crustaceans and frogs is authorised for people who have a valid fishing license, are members of a certified association for fishing and protection of aquatic environments (AAPPMA) and respect the season dates (if applicable) and the legal capture sizes. It should be noted that legally, some species may be considered "non-listed" or even "likely to provoke biological imbalances". In which case, it is prohibited to release live animals to the natural environment or to use them as bait. The transport of living red swamp crayfish requires a written permit (see Chapter 2 for more details on regulations).

Some of these activities may cause more or less severe disturbances for species not targeted by the intervention, which may in turn cause tensions with the people hunting or fishing those species. This aspect must be taken into account when planning interventions.

■ Sterilisation

Sterilisation of bird eggs and the gathering of eggs (spawn) are means to limit the populations of invasive alien animals. Sterilisation is more discreet than shooting or trapping, which explains why it is frequently used in areas where the public is present (see Figure 87). Gathering of amphibian spawn is also used for invasive species such as the American bullfrog (see Figure 88). These techniques require a great amount of time in order to thoroughly cover the site and should be used in conjunction with other methods (shooting, trapping) to ensure maximum effectiveness (see the management project in volume 2, page 201). Sterilisation of bird eggs is regulated by prefectoral orders.

The sterilisation of grown animals is a technique not yet widely used. In continental France, it has been tested on signal crayfish (Duperray, 2010; Basilico et al., 2013). Large males are sterilised and released prior to the reproduction period in order to progressively reduce reproduction rates (see the management project in volume 2, page 139).

Figure 87

© Daniel Serre

Sterilisation of Canada goose eggs.

Figure 88

© SEBB

Gathering American bullfrog spawn.

■ Chemical control

Chemical controls consisting of poisoned bait have been widely used, particularly for invasive rodents. These methods are inexpensive, but can impact non-targeted species and provoke secondary poisonings. They were regulated in 2007. The use of Bromadiolone to limit coypu and muskrat populations was prohibited in 2007 (decree dated 6 April 2007). Rotenone, an organic substance produced naturally by certain tropical plants that is toxic for many cold-blooded animal species, has been used to control invasive alien fish and amphibians. For topmouth gudgeon, following removal of the native fish species from ponds, Rotenone was applied and the native fish were returned. The result was the elimination of the invasive species and increased development of the native species (Britton et al., 2010). However, the product can prove lethal for other species when used in the natural environment, which explains why Rotenone was finally prohibited on 30 April 2011 (notification by the Agriculture and fisheries ministry dated 21 April 2011). Other biocides are also available and are effective against species such as alien crayfish (Poulet, 2014), but the regulations governing their use are complex and subject to a number of European directives and regulations. Before they may be used, special authorisations from the Ecology ministry are required.

Indirect control of animal populations

■ Draining and emptying of ponds and lakes

This method is used for certain invertebrates, fish and amphibians. Prior to draining and emptying, barriers and traps must be set up around the entire water body to make sure the targeted species cannot leave the area and disperse into the nearby environment (see Figure 89). The traps must be checked daily in order to free any native species that might have been captured. Filtering systems must also be operational to avoid the escape of the targeted species. Any remaining animals in puddles of water can be fished or eliminated using lime. If the water body is emptied several consecutive years, the success of the management operation is virtually guaranteed. It should be noted that emptying of water bodies requires an authorisation (see the management project in volume 2, page 158).

Figure 89

Barrier and traps set up for American bullfrogs in the Sologne area.

This method could be tested on molluscs. The observations by Leuven *et al.* (2014) on a section of the Nederrijn River in the Netherlands during a five-day low-flow period in the winter of 2012 revealed a significant reduction in the populations of zebra mussels (*Dreissena polymorpha*) and quagga mussels (*Dreissena rostriformis bugensis*). Over the short period, daily air temperatures ranged from -3.6°C to -7.2°C and daily water temperatures measured 10 centimetres below the surface ranged from 0 to 1.8°C. The densities of the two species dropped to almost zero, then started increasing slowly after six months before returning more or less to the previous densities 18 months after the low-flow period. The authors concluded that changes in water levels under severe winter conditions could serve as a technique to temporarily reduce populations of invasive molluscs. Mollusc populations could take two to three years to return to their former status and the authors recommend work to assess the long-term effects of repetitive interventions on this type of living community. The difficulties of this technique should be noted however, in that it can be used only in environments where the water level can be significantly modified and on the condition that very low temperatures occur at the time of the intervention.

■ Modifications of the environment

Another way to limit the disturbances caused by invasive alien animals is to restore and conserve habitats. It is precisely modifications to habitats that often lead to the regression of native species and the installation of alien species. One means to reduce the likelihood of ecosystem invasions is to avoid creating favourable conditions for the installation and development of IASs. Management of natural areas must adapt to these conditions and take into account the risks of biological invasions, including in urbanised environments. For example, to avoid the installation of Canada and Egyptian geese, techniques such as the creation of planted zones along banks, strips planted with flowers to break up meadows and grassy plots, and the elimination of artificial islands are all much less expensive than putting up fences. However, these management techniques are rarely used in France because they are not well accepted by the public.

Table 9 *Table listing the various control methods for populations of invasive animal species (X = can be used, NA = not applicable, ND = no data). Note that the methods listed as usable are not necessarily effective in all situations.*

Taxonomic group	Trapping	Shooting	Hunting / fishing	Sterilisation	Chemical control	Draining / emptying	Biological control	Modified environment
Invertebrates (crayfish)	X	NA	X	X (male reproducers)	X	X	ND	ND
Fish	X	NA	X	ND	X	X	ND	ND
Amphibians	X	X	NA	X (collect eggs)	X	X	ND	ND
Reptiles	X	X	NA	X (collect eggs)	ND	ND	ND	ND
Birds	NA	X	X	X (eggs)	ND	NA	NA	X
Mammals	X	X	X	ND	X	NA	NA	ND

Exclusion of animal populations

Exclusion of animal populations simply means blocking their establishment on certain sites where they can cause disturbances or damage. These techniques are used in particular when crops have been damaged. They can also be used in complex situations where direct population-control measures are difficult to implement, e.g. in urban areas where the public is present. They serve to reduce the disturbances to a tolerable level, but have no effect on population numbers. They can, however, by used in conjunction with other measures to limit numbers. These techniques are not selective, meaning they can also prohibit the access of non-targeted species.

Physical exclusion

Physical exclusion of IASs consists of installing physical barriers and fences. These systems must correspond to the site conditions and the targeted species. Their height, type, configuration, mesh sizes, etc. must be correctly selected and installed to ensure their effectiveness. Currently, these techniques are used primarily for invasive alien birds and rodents (see Figure 90).

Figure 90

© Parcs Québec

Net set up to avoid the presence of Canada geese on a beach in Québec.

However, they are also effective in slowing and even stopping the progress of alien crayfish in colonising upstream sections of certain rivers. Unfortunately, this solution blocks the upstream migration of many fish species and should be considered only in very specific cases where important crayfish issues are involved such as the presence of native crayfish upstream or the existence of a habitat thought to be favourable for their re-introduction. Finally, this solution may not be used in rivers falling under article L214-17 of the Environmental code (Poulet, 2014).

Repulsion

Repulsion consists of inducing a behavioural change in the targeted species and making it move away from sites where disturbances have occurred. This short-term method is used primarily against birds. Repulsion may be visual (balloons and kites in the form of birds of prey, scarecrows, flags, barricade tape) or consist of noise (noise makers).

Table 10 sums up the advantages and limits of the main techniques used to control invasive animals.

Table 10 *Advantages and limits of the main techniques used to control invasive animals. Adapted from Soubeyran, 2010, according to Courchamp* et al., *2003.*

Control method	Advantages	Limits	Regulations
Trapping	Effective in accessible areas. Selective (category-1 traps).	Often depends on networks of voluntary trappers. Requires a large amount of equipment. Traps can be stolen or damaged. Traps must be checked daily. Trappers must be trained to recognise non-targeted species. Significant trapping skill is required. Special bait must be used. Ethical problems may arise. Selectivity can vary depending on the type of trap. Limited effectiveness for exclusively aquatic species in open terrain (rivers, large lakes, canal networks, etc.). Possible side effects, e.g. stimulated population.	Regulations on pests. Authorisations and certification of trappers.
Shooting	Highly effective and selective. Ethical method.	Access to sites required (private property). Mandatory training of shooters. Need to inform and educate the public. Not possible in all situations (e.g. urban areas or protected sites). Strict safety rules.	Prefectoral authorisation stipulating the time, site, techniques employed and authorised persons.
Hunting / fishing	Done by holders of hunting licenses and fishing permits. Inexpensive. Possible on private property.	Poor results in terms of limiting populations without sufficient reason to target specific species and establishment of a register. May result in population dispersal. Hunters/anglers must be trained to recognise species.	Regulations on hunting, fishing and pests.
Sterilisation	Acceptable method for the general public and possible under difficult conditions. Low technical level.	Searching for eggs and spawn is time consuming. Interventions must be repeated over several consecutive years and include other management techniques. For certain species, a large number of sterile animals must be released to be effective.	Prefectoral authorisations.
Chemical control	Inexpensive and effective. Easy solution.	Non-selective. Requires special authorisation. Need to inform and educate the public.	Mandatory ministerial and/or prefectoral authorisation.
Draining / emptying	Respects the environment and is effective. Low costs.	Requires authorisations from owners of private property. Poorly accepted by pond/lake users. Effective if left dry for several years. Barriers, traps and filter systems must be installed to avoid escapes.	For lakes larger than 0.1 hectare or created by a dam, the decree dated 27 August 1999 applies.
Biological control	Can be implemented in all target zones (e.g. no difficulty in accessing remote areas). Self-sustaining results over the long term. Fewer risks for the environment (no use of biocides or of non-selective techniques). Long-term costs lower than for repeated, standard, management techniques.	Duration and cost of the preliminary research programme to identify, verify and test the potential agents. Time required by the agent, following release, to disperse and provoke the desired effects among the target population. Uncertainty concerning the degree of effects on target population caused by the control agent. Unforeseen impacts caused by the control agent on non-targeted, native species or communities. The biological-control mechanism can reduce densities, but not eliminate the target species.	Regulations on the introduction of biological-control agents (Agriculture ministry).
Modified environment	Preventive and curative approach. Respects the environment. Effective over the long term..	Must be done before developing the site. Poorly accepted by the public.	Regulations depend on the type of site (natural environment, public or private property).
Physical exclusion	Applicable in areas where the public is present and where killing is not possible. Inexpensive (if exclusion systems already exist). Durable over time.	Expensive and complex to set up if exclusion systems do not already exist, require maintenance. Also excludes non-targeted species.	For rivers, check the river classification (EC L214-17).
Repulsion	Applicable in areas where the public is present and where killing is not possible.	Short term only. Can trouble non-targeted species.	No particular regulations.

Biological control of invasive alien species in aquatic environments

This section was drawn and adapted from the second News Bulletin of the Biological invasions in aquatic environments group (see http://www.gt-ibma.eu/activites-du-gt-ibma/lettre-dinformation/lesdossiers-de-la-lettre-dinformation/).

In agriculture, biological control is a technique used to combat a pest or weeds by bringing into play natural organisms that adversely affect the pest or weed, e.g. plant-eating insects, parasitoids, predators and pathogens (viruses, bacteria, fungi, etc.). Following significant developments in agriculture, this technique used to eliminate or regulate pests impacting human activities was expanded to include IASs colonising natural environments. Generally speaking, biological control can be defined as the use of “a living organism to regulate species seen as pests” (Beisel and Lévêque, 2010).

The ecological and economic damage caused by the proliferation of IASs has now been somewhat better assessed, as have the costs of management work intended to repair the damage. The control techniques commonly used in the past (mechanised techniques, plant-protection products, etc.) are expensive, often difficult to implement, not always effective and in some cases they can have negative impacts on the environment. Given current efforts to reduce management costs and improve results, the issue of biological control frequently arises during debates. The method would appear ideal in that it is apparently inexpensive, easy to implement, applicable over large areas and without adverse effects for the environment. But what do we actually know about it? What lessons can be drawn from past experience and what improvements have been made since?

The history of biological control in aquatic environments

Research in this field goes back over a century and even though the example from South Africa presented in the IBMA document addressed only terrestrial plants up until the 1970s, it illustrates the work accomplished and the approaches of that time.

Starting soon after 1900, questions arose concerning the need to study how an invasive species developed in the countries where it was native, whether it was pervasive in those countries, whether natural enemies kept it in check and whether it was possible to import those enemies from the countries in question. A further issue had to do with whether all plants imported to areas where the natural enemies did not exist became pests.

Toward the end of the 1970s, the work in South Africa on aquatic plants addressed a majority of the most troublesome species in the tropical areas of the world, namely water hyacinth (*Eichhornia crassipes*), the topic of the initial work, followed by giant salvinia (*Salvinia molesta*) and water cabbage (*Pistia stratiotes*) at the end of the 1980s, and more recently by parrot-feather watermilfoil (*Myriophyllum aquaticum*) and water fern (*Azolla filliculoides*) (Moran *et al.* 2013).

A great deal of research work was undertaken in the 1970s and in drafting a review of biological control issues for aquatic pests, Schuytema (1977) consulted over 500 documents and discussed all the organisms available for biological control. He also included the possibilities of biomanipulation consisting of modifications to the environmental conditions of species, e.g. the reduction or total deprivation of light, changes in the nutrient content of water, etc., and of interspecific relations, e.g. selecting fish species to control phytoplankton or plants capable of competing with invasive plants. In his report, he noted that a great deal of the listed research consisted of lab work and that there were *"relatively few well documented instances of large-scale control projects"*.

His analysis showed that grazing and predation were the most commonly used techniques, particularly the control of macrophytes by fish. Many of the plant-eating animals and predators do not attack only the targeted species and therefore represent a risk for other organisms in the ecosystem. For this reason, great care must be taken in using them. Single-target insects can be much more effective. According to the review, pathogens were already seen as potentially effective control organisms, but had not yet been used in large-scale control projects. Similarly, biomanipulation was seen by many as a promising set of management techniques.

The review dealt with aquatic plants in particular, including water hyacinth (*Eichhornia crassipes*, see Figure 91). For that species, the review mentioned that *Neochetina* coleoptera were then undergoing an evaluation and were thought to be promising. Since then, these coleoptera (*Neochetina eichhorniae* species) have been extensively used. Beisel and Lévêque (2010) noted that among the one hundred insect species tested for water hyacinth, approximately one dozen "turned out to be capable of inflicting major damage to leaves" and that weevils are used in the United States, Africa and China.

Figure 91

© A.Dutartre, Irstea

*Water hyacinth (*Eichhornia crassipes*).*

Schuytema also mentioned a fungal pathogen (*Uredo eicchorniae*), that was then being studied in Argentina to control water hyacinth. Since that time, at least half a dozen fungal species have been studied and at least one, *Cercospora rodmanii*, is thought to have been successfully tested on water hyacinth.

Potential for Europe?

Since the beginning of these research efforts, worldwide over 7 000 introductions of approximately 2 700 biological-control agents have taken place, primarily in South Africa, Australian, New Zealand and North America (Pratt et al., 2013). In Europe, to date only one biological-control agent has been introduced to control an invasive alien plant, namely the psyllid *Aphalara itadori* (see Figure 92), that was released in the U.K. in 2010 to control Japanese knotweed (*Reynoutria japonica*) (Shaw et al., 2011).

Figure 92

© R. Shaw

The psyllid Aphalara itadori*, a potential biological-control agent for Japanese knotweed.*

In the past, several cases of failed biological control (a recent example being the coccinellid beetle, aka Asian ladybugs) drew significant attention and may be the reason for the reticence in European countries to use biological-control agents against invasive species. That being said, biological control is already used in the overseas territories, notably on Reunion Island (against wild raspberry, *Rubus alceifolius*) and French Polynesia (against miconia), where the results are positive for the time being (Le Bourgeois et al., 2004; Meyer et al., 2007).

Europe is slowly taking an interest in the subject, in order to reduce costs and diversify the methods used to manage invasive plants, but it is limited by the Water framework directive which requires that good ecological status be achieved by 2015, meaning that IAS management must be large scale, but without using phytocide products which are increasingly prohibited in aquatic environments.

In their 2006 review of the potential of biological control for invasive aquatic plants in Europe, André Gassmann and his colleagues at the Centre for Agricultural Bioscience International (CABI) (Gassmann et al., 2006) noted that floating and emergent species such as water fern (*Azolla filliculoides*), least duckweed (*Lemna minuta*), water primrose (*Ludwigia* spp.), water pennywort (*Hydrocotyle ranunculoides*) and New Zealand pigmyweed (*Crassula helmsii*) were "good targets" for standard biological control using host-specific *Chrysomelidae* and *Curculionidae* coleoptera (see Figure 93). They also noted the potential of fungal pathogens against floating and submergent species, and that the use of native pathogens (mycoherbicides) appeared promising.

Figure 93

©S. Reeder

The weevil Stenopelmus rufinasus*, a potential biological-control agent for water fern.*

In the U.K., Japanese knotweed was targeted for the development of a biological-control programme. Annual management costs of the species, known for its impact on biodiversity and river banks, have been estimated at 255 million euros. Standard management (mechanised and manual uprooting) is expensive and must be carried out over many years, yet remains relatively ineffective. The idea of developing biological control was then discussed and a research programme was launched by CABI and its partners in the year 2000 (Pratt et al., 2013).

During the first phase, the natural enemies of the species in Japan were identified and a number were selected for tests on knotweed in areas where it had been introduced. The tests highlighted the high effectiveness of two agents, of which one was the psyllid *Aphalara itadori*, a very host-specific insect. Three years of tests confirmed the specificity of psyllid consumption (90 other native plant were also tested). Following a survey of public opinion and after obtaining the necessary authorisation, the control agent was released in 2010 on a dozen sites in the U.K. The insects survived the winter, but the population numbers were too low to have any significant effect. Another 150 000 insects were released in 2013 and no impacts on native plants and invertebrates were noted. Research is now being conducted on the impact of a mycoherbicide (*Mycosphaerella polygoni-uspidati*), which may be used as an additional biological-control agent.

The results of this initial experiment are not yet available, but CABI has already drawn up a number of recommendations on how to run a biological-control programme (Shaw et al., 2011).

■ Take care in selecting the target plant, notably in terms of its sensitivity to biological control, but also taking into account public opinion as well as the economic and political aspects.

■ Use the existing legislation concerning plant health and protection, notably when analysing the risks of the biological-control agents and in order to obtain, in a completely legal manner, the authorisations to import, transport and release the agents to the environment.

■ Draw up a list of plants on which the biological-control agent will be tested (safety procedure). This list should include economically significant plant species and take into account public opinion. The list should also be confirmed well in advance of the test phase.

■ Prepare a monitoring plan before releasing the agent in order to detect any unforeseen impacts on the environment. This plan should be designed and funded for at least five years, cover several sites and include safety measures (insecticides and herbicides if native species are threatened).

■ Provide ample information to the public, including clear messages on the objectives of a biological-control programme (a reduction in the numbers of the target species to below a tolerable threshold, but not total elimination) and on how the programme will be carried out, and answers to frequently asked questions, e.g. what will the insects eat once the knotweed has been consumed, what about the cases where biological control "failed" (cane toads in Australia, Asian ladybugs in Europe), etc.

This type of programme requires a very high research budget that, generally speaking, only national and international organisations can muster. Programmes often last over a decade, an example being the programme against Japanese knotweed. This time is required to run the tests to check the specificity of the control agent with respect to the living communities into which it will be introduced.

Box 25 presents the generally accepted steps in the procedure leading to a management decision concerning a species seen as sufficiently troublesome to justify the release of an effective, host-specific agent.

Box 25

The steps in a biological-control programme for an invasive alien plant

1. Launch of the biological-control programme (selection of the targeted invasive plant, analysis of any conflicting interests, bibliographical review of the targeted plant and of its natural enemies).
2. Research and monitoring of the introduction area (identification of any natural enemies for the targeted host, check that no local, effective control agent exists).
3. Work abroad in conjunction with the research organisations in the native region of the target species, research on and monitoring of any natural enemies, prioritisation of the species demonstrating high potential as a control agent, obtaining the necessary authorisations to monitor and export the agents.
4. Study the ecology of the target species and its natural enemies. Compare the ecology of the species in its native range and in the introduction area, study the climatic and ecological conditions required for the development of the biological-control agent.
5. Study the specificity of the biological-control agent. Assess in the lab and in the field the physical, chemical and nutritional factors that will determine the specificity of the control agent's consumption, run tests on a wide range of native species (the test plants).
6. Release the agent to the environment and monitor it. Once all the scientific studies have been conducted, draft the report file for the oversight authorities including the monitoring programme and an analysis of the risks.

Advantages and unknowns

Similar to any other management technique, biological control has a number of advantages over the other techniques as well as a number of inherent unknowns or risks (see Table 11). These unknowns or risks have to do with our insufficient knowledge on the species, both those to be controlled and those to be introduced, as well as on the ecological functioning of ecosystems, both before and after the introduction.

Table 11 *Advantages and unknowns of biological control. According to Shaw* et al.*, 2011.*

Advantages
■ Can be implemented in all target zones (e.g. no difficulty in accessing remote areas) ■ Self-sustaining results over the long term ■ Fewer risks for the environment (no use of biocides or of non-selective techniques) ■ Long-term costs lower than for repeated, standard, management techniques
Unknowns
■ Duration and cost of the preliminary research programme to identify, verify and test the potential agents ■ Time required by the agent, following release, to disperse and provoke the desired effects among the target population ■ Uncertainty concerning the degree of effects on the target population caused by the control agent ■ Unforeseen impacts caused by the control agent on non-targeted, native species or communities ■ Numerous administrative authorisations required (import, breeding and release of control agents) and difficult to obtain

Economic factors are an important aspect of the studies on biological control, representing much higher initial budgets than other techniques (roughly speaking, several years of research compared to the purchase of equipment), but subsequently lower costs because the agent continues to produce an effect whereas equipment is a source of continuous costs. The fact that a degree of uncertainty remains concerning the later development of the management programme is not a sufficient reason to cancel the programme, but means that the programme should be monitored to keep an eye on its status and to gather knowledge on the species and ecosystems in question. A further element of uncertainty, in addition to all the others, lies in the increasingly perceptible effects of climate change.

If we accept the wider definition of "biological control" from Schuytema (1977), grazing is also one of the management techniques. For at least 20 years, extensive grazing has been a technique commonly implemented for emergent and amphibious plants in wetlands located on protected sites (nature reserves, hunting reserves, etc.), using either local breeds of animals suited to wetland conditions or imported species. That is why the silhouette and the large, upraised horns of Highland cattle, a rustic breed of cows from Scotland, are now well known to the visitors of many nature reserves in continental France where the animals consume most of the plants in those habitats.

Occasionally, cows or horses have been observed eating the water primrose that had colonised the grazed sites. In at least one case, a local breed of cows is thought to have eaten some of the water primrose along a lake, but that was not observed in other cases. In the Barthes de l'Adour area, horses were placed on a site heavily colonised by water primrose, but refused to eat the plants and had to be removed before they starved. The wide discrepancies between these observations and the absence of monitoring protocols means it is not possible to draw any conclusions concerning "extensive" management.

Unfortunately, experiments on "intensive" grazing, undertaken with monitoring protocols using herbivores targeting a specific plant species, did not produce any clear results. For example, a test conducted according to a precise protocol in the Barthes de l'Adour area, with buffaloes, animals with a reputation of being very effective herbivores, did not produce the desired results, i.e. the animals did not eat the water primrose. On the other hand, an experiment, using domestic goats (*Capra aegagrus hircus*) (see Figure 95) to eat knotweed, produced excellent results (see the management project in volume 2, page 94).

Figure 94

© CG 53

*An experiment using domestic goats (*Capra aegagrus hircus*) to browse knotweed in the Mayenne department.*

The use of sheep to control invasive, terrestrial plants, often in urban and periurban areas, is now increasingly mentioned in the media, including in North America where the animals and their consumption habits are presented as an alternative to herbicides (see for example http://www.beyondpesticides.org/dailynewsblog/?p=11473).

In addition to the issue of the monitoring protocol used to determine the effectiveness of this technique, one of the main difficulties lies in monitoring the introduced animals, particularly in terms of their health and the evolution of the population. It would be necessary to run the tests over long periods in order to fully judge the effectiveness of this technique, unfortunately this is not possible given that most experiments are relatively short. The need for regular monitoring is probably the main disadvantage of this technique because unforeseen events can occur, e.g. the death of animals (see the management project in volume 2, page 94).

The unforeseen events are occasionally very surprising. Certain aquatic birds, very common in continental France, are effective herbivores, e.g. swans, red-crested pochards (*Netta rufina*) and Aythya ducks (*Aythya* spp.). The joint assessment conducted by INRA and Cemagref in 1981 (Dutartre et al., 1981) resulted in a proposal to test ducks and swans as a means to control hydrophytes in a small lake that had been totally colonised by the plants. The lake, less than two hectares in size and located on the premises of a school where the birds could be regularly monitored by a competent person, appeared to be well suited to the test. The monitoring protocol stipulated the presence of areas where the birds could not enter to eat the plants. A dozen couples, including three couples of swans, were established on the lake. According to the monitoring results, the birds were effective (the criterion being no notable development of the hydrophytes) for the first two years, then the plants started developing again and subsequently the situation rapidly worsened (Dutartre and Dubois, 1986). The main cause of the failure was the instability of the bird population over time, due to the arrival of outside birds and in-breeding with mallard ducks in spite of the regular monitoring.

For a long time, grass carps (*Ctenopharyngodon idella* Val.), aka the white amur, were one of the plant-eating fish most commonly mentioned as a biological control for aquatic macrophytes, initially in tropical zones and then in temperate zones. They have been present in Europe for approximately 30 years. Their introduction in France is prohibited, however their story illustrates not only how human needs, perceptions and opinions can change over time, in step with events, but also the new knowledge gained and the difficulties in sharing information (see Box 26 and Figure 95).

Box 26

Herbivorous carps (grass carps)

The grass carp (*Ctenopharyngodon idella*)[10], also known as the white amur, is a plant-eating fish with an undeniable taste for aquatic plants and, if the fish has exhausted the aquatic resources, even for the leaves of plants growing along the banks that dip into the water, to the point that it has often been called a mowing fish. Though its capacity to consume plants is undeniable, the fish shows no real appetite at temperatures below 15°C, which means it is much less effective in cooler waters. The incomplete digestion of the consumed plant material means that the fish releases to the water significant quantities of organic matter and the degradation of this waste can create oxygenation problems in smaller, stagnant water bodies.

What is more, grass carps are picky about what they choose to eat. This led, some 20 years ago, to a study on the subject (Codhant and Dutartre, 1992) which showed that, at least in the lakes in the Landes department, the plants preferred by grass carps were not waterweed and water primrose, but rather native species such as watermilfoil and pondweeds, which was not the desired outcome! These food choices are obviously one of the limitations weighing on this plant-eating species because they can substantially hinder the introduction of the fish in environments where the native plant communities are ecologically important or, more generally speaking, in aquatic environments used for multiple purposes where the native communities often serve to protect the multiple uses.

*10. Not to be confused with silver carp (*Hypophthalmichthys mollitrix*) or bighead carp (*Aristichthys nobilis*), two species that feed preferably on zooplankton and phytoplankton.*

In their review of the ecological impacts of grass carps, Dibble and Kovalenko (2009) noted that in lakes and ponds (i.e. in the absence of reproduction), grass carps can degrade water quality. The stirring of sediment during feeding and the decomposition of its digestive waste produce an increase in the concentrations of nitrites, nitrates and phosphates, followed by a drop in dissolved oxygen and, often, by algal blooms. These changes may persist over the long term and even become irreversible. Adverse impacts were also noted for the communities of aquatic plants, macroinvertebrates and fish. Given the complexity of the interactions and the lack of study on the ecological mechanisms involved when grass carps are released in an ecosystem, it is very difficult to foresee the consequences (Dibble and Kovalenko, 2009). Consequently, caution is advised before introducing this species into environments where protected species are present.

Starting in the 1970s, various publications highlighted the plant-eating capacities of the species and the review by Schuytema (1977) did not neglect this aspect. One of the arguments used to promote the species was the fact that it was possible to control it, given that it could not reproduce in the new environments. The species requires the current conditions prevailing in Asian rivers, from where it came, in which its pelagic eggs can develop through their complete cycle until they become alevins. However, efforts to reproduce the fish under controlled conditions succeeded, which made it possible to produce the species commercially in Europe and North America.

As a result, this species could demonstrate its “mowing” capabilities in many ponds and lakes. Then, approximately a decade later, researchers in the United States started to wonder why the populations of certain fish species were falling in large lakes. They discovered that the massive introductions of grass carps had eliminated the plant beds and the corresponding fauna used by the other fish species for laying their eggs and/or as a source of food.

A further surprise arrived when natural reproduction of the species was observed in the Mississippi. It would appear that grass carp succeeded in adapting to the ecological conditions of at least one of the major rivers in North America, a feat previously considered impossible. In continental France, according to a study conducted by the University of Nancy, only adult fish have been observed or occasionally captured (approximately 20 per year) in rivers (Teletchea and Le Doré, 2011). This lack of reproduction in European rivers is still today an argument for the sale of this fish in France and numerous requests to market the fish have been made by fish farms. Currently, it is possible to introduce grass carps into the natural environment, but only in “closed” water bodies and with a prefectoral authorisation (decree dated 20 March 2013).

Figure 95

© USGS

Ctenopharyngodon idella.

Concerning invasive alien fauna, a few attempts have been made in continental France to introduce native predators, e.g. the introduction of eels to control red swamp crayfish in the Brière marshes (see the management project in volume 2, page 129), but these efforts remain relatively marginal. The available knowledge on the predation by carnivorous fish of invasive animals (crayfish, African clawed frogs, etc.) has raised doubts concerning the usefulness and the effectiveness of these introductions. The characteristics of the targeted prey (size, ease of consumption, etc.) and the presence of non-targeted, protected native prey such as amphibians make introductions in the natural environment a complex and risky undertaking.

Management of waste produced by interventions on invasive alien species

The regulations governing the management of waste produced by management interventions on IASs is presented in detail in Chapter 2.

One of the difficulties of management work that managers have systematically encountered since the first interventions on invasive species concerns the waste produced. What should be done with the plants and animals that are withdrawn from sites, occasionally in very large quantities?

Organic transformation of invasive alien plants

Given that a large number of interventions on invasive alien plants take place each year, the future of the resulting organic matter must be foreseen as an integral part of the management system. This aspect was long neglected and as a result no generally applicable solutions were developed, the waste was often simply deposited nearby or in a landfill. In a few cases, the plants were spread directly in fields where farmers first let them dry, then ploughed them under. The increase in the quantity of plant waste and changes in the regulations governing the management of green waste made it necessary to reassess the problem as a whole and to change work habits (Dutartre and Fare, 2002).

A number of methods were extensively used, each with a set of disadvantages and consequences that were analysed in view of progressively selecting those methods comprising the least risks, short term and long term, for the environment. For example, burying the plants near aquatic environments and burning them after drying were two techniques used in an array of situations, but the difficulties involved (excavation work, destabilisation of soil, etc. for the first, safety concerns for the second) led to a progressive halt in their use.

Depositing the plants on the banks, either spread or in piles, was a much more widely used technique (see Figure 96). In general, this technique did not cause any particular problems when the waste was made up of submergent plants that dry quickly and do not regrow. Problems arose for amphibious and terrestrial plants that better resist drying and are capable in some cases of striking roots or of surviving at least one or two years on piles where the decomposition of the middle of the pile enables the plants on top to survive. For these reasons, implementation of this technique as well dropped significantly.

Figure 96

Parrot-feather watermilfoil deposited on the banks of the Léon Pond (Landes department).

In parallel, a number of studies addressed the agricultural and forestry use of water primrose waste and that of certain submergent species. For example, for several years water primrose has been spread under trees in the Landes or on dry agricultural land prior to being turned under near the Marais Poitevin marshes[11]. Large quantities of waterweed extracted from the Dordogne River have been mixed into the green waste of the Bergerac urban area and composted. The primary objective is to dispose of the waste produced by interventions in a manner avoiding the environmental risks involved in depositing the waste in natural areas. The reuse of the waste is a secondary issue that nonetheless facilitates the overall approach given its positive aspects.

Studies on management methods for invasive-plant waste continued over the years and dealt increasingly with the possibilities of reusing the large quantities of organic matter.

Though it may be tempting to see invasive plants as a form of ultimate waste (see Box 27) because they are difficult to treat given the risks of dispersal, years of experimental work have proven that the organic matter can be put to use. There is no reason to put this waste in a waste-storage centre or to send it to a household-waste incineration centre. Once they have been withdrawn from the natural environment, invasive plants are a form of green waste (see Box 27) that should be processed in a manner limiting the emission of greenhouse gasses and that returns the organic matter to the earth. According to the ministerial circular dated 10 January 2012 concerning on-site sorting of biowaste (see Box 27) by large producers (see the Grenelle 2 law), the two available techniques are composting and methanisation. They produce an organic fertiliser that can be directly used in soil, i.e. compost and digestate (methanisation residue), where the second can also be transformed into compost.

Depending on the type of plants, the quantities harvested and the site location, it may be worthwhile to ship the waste to an industrial processing plant (see Box 27) for processing under controlled conditions. The reason is that optimum composting conditions, notably in terms of the temperature, cannot be maintained in rudimentary systems. It should be noted, however, that not all processing plants offer the same technical conditions and they may be more or less equipped to process this type of waste. It is preferable to avoid any intermediate steps, e.g. depositing the waste in a dump, and to ship the waste directly to the processing plant, thus reducing any risks of dispersal and the costs.

*11. A technical report is available at:
http://www.pays-de-la-loire.developpementdurable.gouv.fr/IMG/pdf/Proposition_d_une_methode_de_recyclage_et_de_valorisation_agronomique_des_jussies_extraites_des_milieux_aquatiques_cle5d3578.pdf.*

Definitions for the waste of invasive alien plants

■ According to article L541-1 in the Environmental code, “waste, whether or not the product of processing of other waste, is considered ultimate, in the sense provided for in this chapter, when it cannot be processed using the technical and economic means available at the time, notably by extracting the useful part or by reducing the danger and/or the polluting nature of the waste”.

■ Green waste is organic waste produced by cutting grass, trimming hedges and bushes, cutting branches, clearing land and other similar activities (ministerial circular dated 18 November 2011 on prohibiting open-air burning).

■ Biowaste is any non-dangerous, biodegradable waste from gardens and parks, any non-dangerous kitchen waste or food, notably produced by households, restaurants, caterers and retail stores, as well as any other comparable waste produced by companies producing or transforming food.

■ Methanisation plants and composting units are professional facilities listed according to the Regulated installations for environmental protection (ICPE) criteria because they may be dangerous or create problems for the neighbours, for public health and safety, for agriculture, for nature, the environment and landscapes, for rational energy use or for special sites, monuments and the architectural heritage.

■ Hygienisation is a process employing physical and chemical means to reduce to a non-detectable level the presence of pathogenic micro-organisms in an environment (decree dated 8 January 1998 on the spreading of WWTP sludge, articles 12 and 16).

■ Industrial composting units

Composting principles

Contrary to anaerobic methanisation, composting is an aerobic process (with oxygen) for fermenting matter under controlled conditions. The organic matter may be similar in nature or a mix of different types of feedstock. In the latter case, one speaks of co-composting, i.e. a combination of biowaste and/or WWTP sludge, livestock effluents, waste from food industries, etc.

Composting produces CO_2, heat and an organic residue with a high humus content, namely compost. The high temperature during composting, greater than 55°C to 60°C over several consecutive days, results in hygienisation of the final product. In some cases the temperature can reach 80°C and care must be taken to avoid fires. The rise in temperature takes place during the fermentation when the most easily degradable matter decomposes. Fermentation is followed by a maturation phase that stabilises the compost and removes any phytotoxicity. The temperature then drops, producing the humic compounds.

Prior to processing, the waste can be shredded to facilitate its degrading and, depending on the types of waste, mixed. The waste is then arranged in large mounds. During fermentation, frequent mixing or managed aeration may be required to achieve accelerated processing (for slow composting, the mounds are mixed less often, generally once per month). Between the fermentation and maturation phases, or following the maturation phase, the compost can be sifted to sort the different grain sizes depending on the subsequent uses.

Depending on the process employed, complete composting may take from four to six months.

The different types of composting units

There are three types.

- Local governments, under their own or external management, can make use of the green waste and/or biowaste produced in their area (see Figure 97). They are often not interested in receiving waste from other areas because their facilities are generally sized precisely for the foreseeable quantities from their local area.
- Private companies see compost production as a profitable activity. The resulting product is sold to farmers, landscape professionals and the general public.
- Farmers often undertake co-composting, mixing green waste (from local governments, the public, companies or from their own farm) and their agricultural waste (livestock effluents, crop residues, etc.).

Figure 97

Composting unit.

Methanisation plants

Methanisation principles

Methanisation is a natural, biological process that degrades organic matter due to the combined action of different anaerobic bacteria (bacteria that develop in the absence of oxygen). The feedstock is placed in a tank, the digester, then heated and mixed for a period of 40 to 60 days. The process produces the digestate, a fertiliser in the form of a viscous residue, and biogas, which is a renewable energy. The digestate may be spread in compliance with a spreading plan as is or following separation of the liquid and solid phases. The biogas is composed primarily of methane. It can be vaporised during cogeneration (combined production of heat and electricity) or injected, following purification, into gas-distribution networks.

The different methanisation techniques

There are different types of methanisation units. They are often created by a group of farmers wishing to make use of their livestock effluents and crop residues, in which case one speaks of farm methanisation units. Larger, collective methanisation units can receive waste from a wider area, in which case one speaks of territorial units. Whatever the type, they select a methanisation technique suited to their specific needs. There are two main methanisation techniques.

- The continuous, liquid process (currently the most common) involves complete mixing, where the digester receives a daily “ration” of organic matter in which dry matter must not exceed 18%.
- The discontinuous, dry process where at least four digesters mounted in parallel and operating simultaneously are supplied on different days (e.g. each digester every ten days). A particular feature of this dry process is that the digesters can receive larger waste in which the dry matter can extend well beyond 25%.

Depending on the process selected, more or less fresh plant waste can be used. In the liquid process, mixing of the plant waste with other waste in the daily ration is a means to create a degree of flexibility. It is essential, however, that the waste not have started to ferment during prior storage because the quantity of available methane is reduced.

The temperature produced during the process is a critical factor for IAS waste. If there is a risk of seeds being present in the IAS waste, it is necessary to exceed a temperature of 50°C to eliminate the germination capacity of water-primrose seeds and 60°C for knotweed. Certain facilities attempt to produce an optimum quantity of biogas in a limited amount of time. This type of process, called themophilic, can reach temperatures between 48°C and 60°C. However, the most common processes are mesophilic, i.e. they operate at a temperature of around 38°C, the ideal temperature for the bacteria.

The necessary conditions for effective methanisation

Woody debris cannot be used in the digester because the bacteria involved are incapable of degrading it. For this reason, it is preferable to use aquatic and amphibious plants that have low or no ligneous content. For the liquid methanisation process, it must be easy to pump the organic matter, i.e. it must have been previously shredded (10 cm maximum in size) to facilitate handling. In that not all methanisation units are equipped with shredders, it may be necessary to plan on shredding the waste before shipping it. Finally, similar to all other methanisation waste, it must not contain any inert material (sand, gravel, glass, plastic) that could alter the methanisation process by provoking sedimentation, phase separation or surface deposits. This explains why waste produced by mowing is preferable to that produced by uprooting.

The daily ration input into the digester is adjusted as a function of the methane-producing capacity of the various types of waste used (animal waste including liquid and dry manure, bird droppings, crop residues, food-industry waste, waste from local governments including biowaste, WWTP sludges and greases, cut grass) and it is possible to modify the ration if the arrival of the IAS waste can be planned a few weeks in advance. Unfortunately, the methane-producing capacity is not precisely known for all aquatic and amphibious invasive species of plants and this constitutes a limitation in efforts to adapt their treatment. Finally, methanisation units rarely dispose of the necessary space to store feedstock not included in their normal production schedules, consequently the timing of the arrival of invasive-plant waste for use in the digester must be carefully planned.

■ Selecting a process for the waste of invasive alien plants

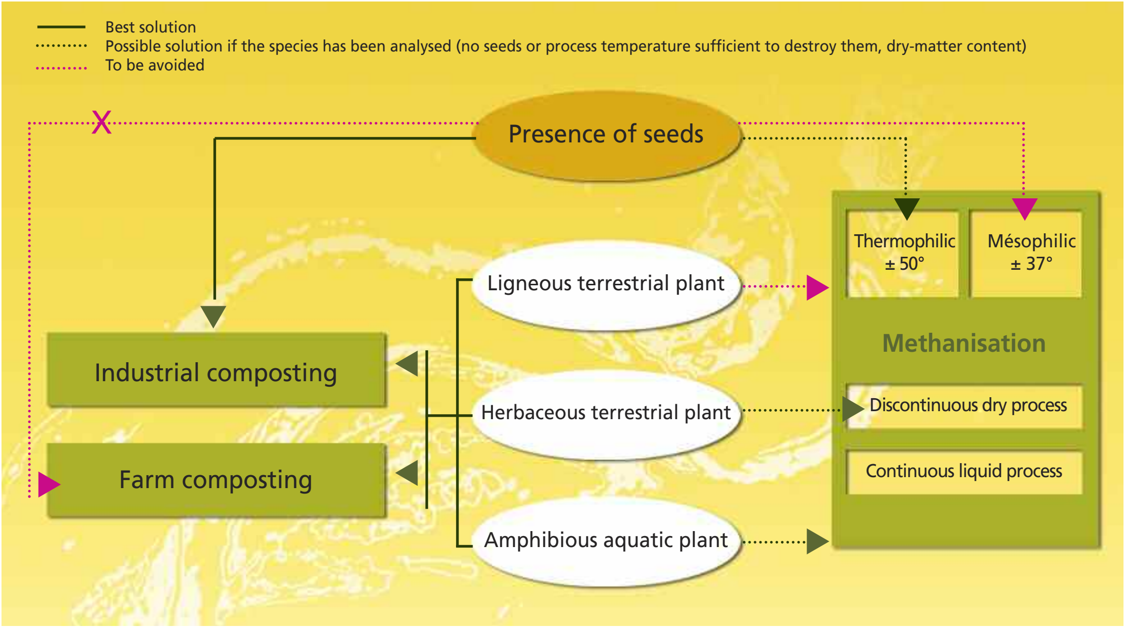
Figure 98 indicates how to select the best process (composting or methanisation) depending on the type of waste and the presence or absence of seeds. Not all composting centres and methanisation units accept invasive-plant waste and it is necessary to contact them first in order to discuss the processing possibilities.

■ The limits to these solutions

Increase in management costs

Processing invasive-plant waste to transform it into a useful product represents an additional cost that the manager must take into account. The first step of transporting the waste away from the intervention site is one part of the cost, whether it is carried out by the manager or by an outside supplier. Some processing centres may be in a position to organise the transport or to provide containers. The cost of processing depends on the pricing policy of each centre. Some do not require any payment, particularly methanisation units if the waste is known to have high methane-producing capacity, whereas others set their price per ton depending on the constraints weighing on the type of waste. In general, prices are set on a case-by-case basis.

Figure 98



Selection of the best process depending on the type of waste and the presence or absence of seeds.

Shifting from waste to valuable product

Processing of organic waste in view of creating a product results in the production of compost or of digestate and biogas. Compost is covered by standards[12] and is a valued fertiliser that can be freely marketed. Digestate, on the other hand, is considered a waste product that may be spread only under highly regulated conditions. There are two ways to turn digestate into a product. The first is via a certification procedure (lasting 12 to 18 months and costing approximately 40 000 euros) to prove its agricultural value and its innocuity, the second is to use it in a composting centre to produce compost. The biogas can be used to produce electricity that is then sold to EDF (the national electricity company). In this manner, a valuable shift can be achieved when processing aims not only to eliminate the waste, but to create a marketable product.

Unfortunately, that may be difficult for two reasons. First of all, invasive plants are a seasonal source of waste. Consequently, the processing centres cannot count on this source for a regular supply. The processing centres may also be reticent to accept invasive plants if they are unfamiliar with how they react during composting or methanisation. Centres aiming to create a marketable product will not take the risk of reducing the performance level of their facilities by incorporating invasive plants.

■ Possible futures?

Invasive plants and their waste need to be managed and current research for solutions aims to develop processing techniques, either specifically for invasive-plant waste or by adapting it to the existing processes, while limiting the geographic scale of operations and transportation distances, thus reducing the cost of this indispensable, final phase of management for invasive plants.

Selection of the "short" recycling process (composting) must take into account all aspects of the local situation, i.e. the type of plant (including the germination capacity of seeds) and environment, the intervention objectives, the type of waste, the possibilities for transportation, storage and recycling, etc. Though it will not pay for an intervention, recycling may be a means to limit the overall cost of the work.

12. Standard NFU 44-051 for compost made of plant and animal waste and for urban compost made of household waste, or standard NFU 44-095 for compost made of WWTP sludge.

Management of animal waste

Management work for invasive alien animals can produce relatively large quantities of waste that must be eliminated. To provide a general idea, interventions against invasive rodents can produce over 50 tons of dead animals in certain departments in France (FEVILDEC, 2014). Health standards require that processing of this waste comply with certain regulations. European regulation 1069/2009 addresses the problem of animal waste. For wild animals drawn from the natural environment, which is the case for interventions on invasive species, the regulation applies only to animals suspected of being infected by a transmissible disease.

The Rural code (articles L226-1 to 226-9) lists the requirements for the management of “animal waste”. It is necessary to distinguish between two categories of animal waste, the dead bodies of wild animals and animal by-products. Following an intervention on invasive alien animals, the dead bodies constitute the waste.

The bodies are the responsibility of the public rendering service. The rules stipulate that if the animals weigh less than 40 kilogrammes, they may be buried on site if the land owner agrees. If they weigh over 40 kg, they must be handled by the rendering service.

■ Rendering

Removal is free of charge if the animals weigh more than 40 kg and it is possible to freeze smaller animals in order to reach the 40 kg threshold. Town officials must make a request to the rendering service and set up a pick-up service for the bodies of wild animals as well as a temporary storage system. Departmental collection plans have been established by towns in order to organise and rationalise the collection of animal bodies as well as dispose of them in compliance with the applicable regulations. Approved equipment must be used, e.g. rendering containers, freezers, special plastic and paper bags, etc.). In some departments, towns have created certified collection points that are georeferenced to facilitate their use. It is necessary to contact each town for more information before launching a management intervention.

■ On-site burial

If the animal waste produced by the management work does not exceed 40 kg, the waste may be buried on site. A ditch should be dug in compliance with the following recommendations (Fédération des chasseurs du Languedoc-Roussillon (LR hunting federation), 2010):

- burial with the permission of the land owner;
- on terrain sloping less than 7% (4°);
- outside of wetlands, floodable areas and protection perimeters for drinking water;
- more than 100 metres from a river, lake or abstraction for household use;
- more than 200 m from homes;
- more than 50 m from a road, path or trail;
- more than 50 m from farm (livestock) buildings.

The waste must be covered with quicklime (equivalent to 10 to 25% of the waste weight or one-quarter of the waste volume). The ditch must be deep enough (1.3 metres deep for the largest bodies) and access for animals must be blocked (fence) if possible.

Assessment of interventions

Given the human, technical and financial investment of an intervention and the expected results, a two-pronged assessment is required.

■ The first must determine the actual effectiveness of the intervention with respect to the planned results. It is generally based on relatively simple observations and data collection following the intervention. The environmental managers themselves can often conduct the assessment or provide the information using a pre-established protocol. Various methods have already been used to conduct this assessment which is essentially a comparison, before and after the intervention, of certain parameters selected according to the type of species and the environment.

■ The second deals with the ecological impacts that may be directly attributed to the intervention. This assessment is much more complex because it requires specific monitoring procedures that the managers can generally not implement themselves. This monitoring, awarded to an external supplier such as a consulting firm or a research lab, fills out the first assessment and requires additional funding that may be difficult to obtain, which explains why these assessments are carried out relatively infrequently. This is probably one of the important aspects that must be improved in the coming years in order to more precisely determine the impacts and include them in the work to enhance IAS management. This would hopefully make it possible to reduce the damage to biodiversity caused by interventions. It would be a very positive step forward if this second assessment were progressively included in the planning for management interventions, this providing the information required to make decisions with a clear idea of the management issues and impacts.

Assessment of intervention effectiveness

If it is to serve as the basis to analyse interventions and their results, this assessment must include information on how the intervention was carried out, indicating at least the dates and the duration, the site, the type of environment, the equipment and methods used, and the number of people involved. The above information is required for all interventions, for all types of species.

However, other information on the specific species, fauna or flora, is also required. For plants, quantitative data alone are sufficient, for example the surface area or the linear distance treated, the relative abundance of the plant beds over the entire site or on specific geographically identified plots, the weight or the volume of the plants harvested on the site or from each plot. For animals, quantitative data, e.g. the number of animals of the targeted species removed from the site, their total weight, etc., and qualitative data, e.g. the reproductive stage of the animals, are required.

Various observation sheets and site-report sheets have been devised and used for approximately 15 years, an example being the site-report sheet used by the Loire-Bretagne work group (Haury et al., 2010) (the sheet is available on the www.gt-ibma.eu site). In addition to the general information on interventions listed above, the sheet includes information on the entity doing the work, the cost of the work and on the recycling technique and cost for the harvested plants.

Subsequent analysis of the sheets provides information on how interventions are conducted, on their effectiveness when two or more sheets are available for successive interventions on the same site, and information of use for the economic analyses that are now increasingly carried out, for example the analysis by Matrat *et al.* (2011) presented during the symposium titled "Invasive plants in the Pays de la Loire region".

(http://www.pays-de-la-loire.developpement-durable.gouv.fr/2011-colloque-regional-les-plantes-a1338.html).

Other, more specific sheets for a given territory or type of plant are also available. That is the case, for example, of a recent sheet prepared by the Vendée departmental council, the Vendée fishing federation and IISBN, that concerns a small number of terrestrial and "near-water" plants such as knotweed, groundsel bushes and common ragweed. The sheet, used to "report on the study and/or monitoring of invasive terrestrial plants", serves to collect information on the location and the type of environment in the wet marshes, a description of the colonisation and information on any management work done.

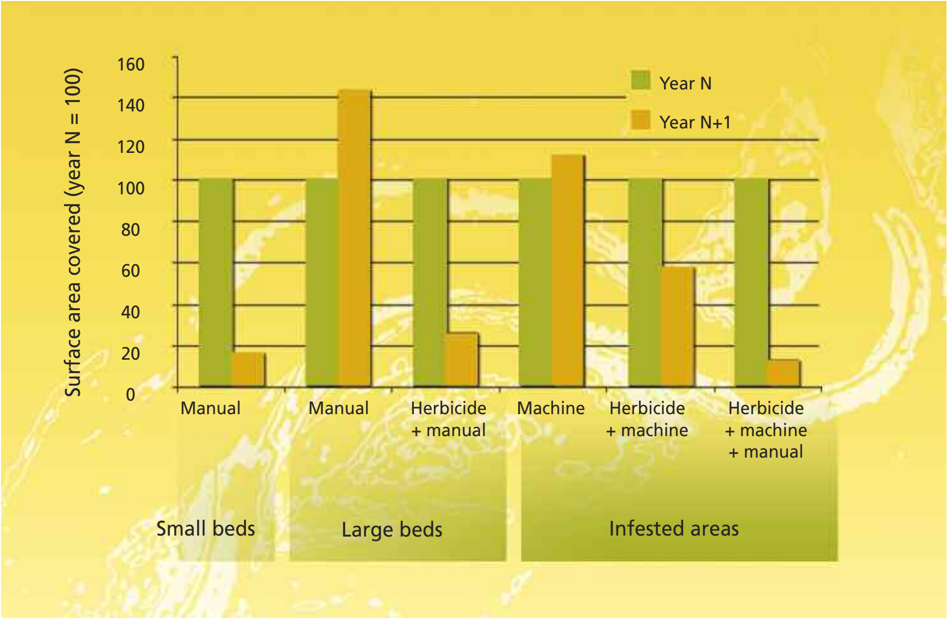
(http://www.sevre-niortaise.fr/accueil/des-thematiques-du-bassin-versant/les-plantes-exotiquesenvahissantes/).

For both plants and animals, the "before and after" comparison can be carried out either using each parameter separately or by creating indicators combining two or more parameters.

For plants, for example, the assessment can be carried out fairly easily by measuring any changes in the surface areas on a site or section of the site. The results of experiments carried out by IIBSN to devise the best management strategy for water primrose in the Marais Poitevin marshes (Pipet, 2007) were analysed in this manner.

The experiments tested, either alone or in combination, three intervention techniques that at that time appeared feasible for wet marshes, namely manual uprooting (manual), mechanised uprooting (machine) and the use of herbicides (herbicide). Effectiveness was assessed by calculating the surface area covered by water primrose the year following the work (year N+1), given that the surface area prior to the work was assigned a value of 100. Figure 99 shows that manual uprooting alone is sufficient for small plant beds, that for large beds it is first necessary to use a herbicide, and that for heavily infested areas, the three techniques must be successively used to obtain the best results, i.e. a value of approximately 15 for year N+1. The results of these experiments convinced the IIBSN to implement the combination of three techniques for at least as long as the herbicides remained effective against the emergent water-primrose leaves (Pipet and Dutartre, 2014).

Figure 99

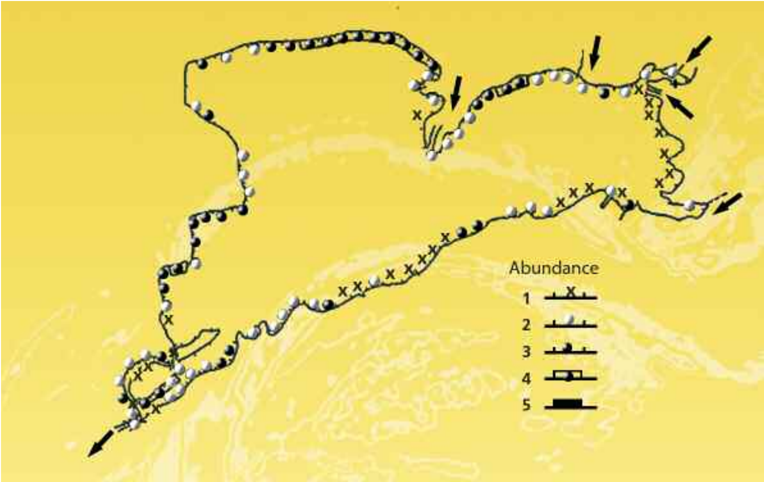


Effectiveness of management techniques measured as a function of surface areas covered by water primrose in the Marais Poitevin marshes. According to Pipet and Dutartre, 2014.

Another assessment possibility is to compare from one year to the next the relative abundance of the species in specific bank sectors of a river or lake. This was the method used in studies and monitoring of colonisation and for the management of invasive aquatic species in lakes and ponds in the Landes department (Dutartre et al., 1989). An assessment of plant abundance (ranging from 1 to 5, i.e. from "very rare" to "very abundant") was conducted in each sector, ranging from 50 to 100 metres long depending on the water body, but identical within a given water body. This system made it possible to identify the most heavily colonised sections and to observe the intervention results the following year, sector by sector.

The sectors were initially marked on a map (scale 1 : 10 000) and subsequently using a GPS. Comparisons of the degree of colonisation were thus possible over time, species by species and sector by sector. An example is provided in Figure 100, with a map showing the locations of large-flower water primrose in the Aureilhan Pond (Landes department).

Figure 100



Assessment of water-primrose abundance in lakes and ponds in the Landes department. Example of the Aureilhan Pond. According to Dutartre et al.*, 1989.*

For animals, the most easily accessible data generally consist of capture reports. The data may cover an entire territory (see the management project in volume 2, page 211) or provide more detailed information on the number of animals trapped or killed per segment of territory. Table 12 presents the number of coypus and muskrats trapped per kilometre of river in the Basse-Normandie region. The authors note that numbers greater than 15 to 20 per kilometre correspond to high densities ("infestations") (FDGDON Manche, 2007). Here again, comparisons from one year to the next provide an idea on the effectiveness of management work.

Table 12

The number of coypus and muskrats trapped per kilometre of river in the Basse-Normandie region. According to FDGDON Manche, 2007.

Department	River basin	Number of coypus per km	Number of muskrats per km
Orne	Upstream Orne	42	5
	Risle	20	11
	Huisne	11	1
	Varenne	14	8
Calvados	Downstream Orne	6	7
	Drives	10	17
	Touques	10	8
	Seuilles	3	15
	Aure	20	36
	Vire	10	14
Manche	Saire	10	12
	Divette	3	9
	Douve	14	7
	Taute	10	13
	Ay	6	9
	Sienne	29	NC
	Sée	3	7
	Séline	7	7

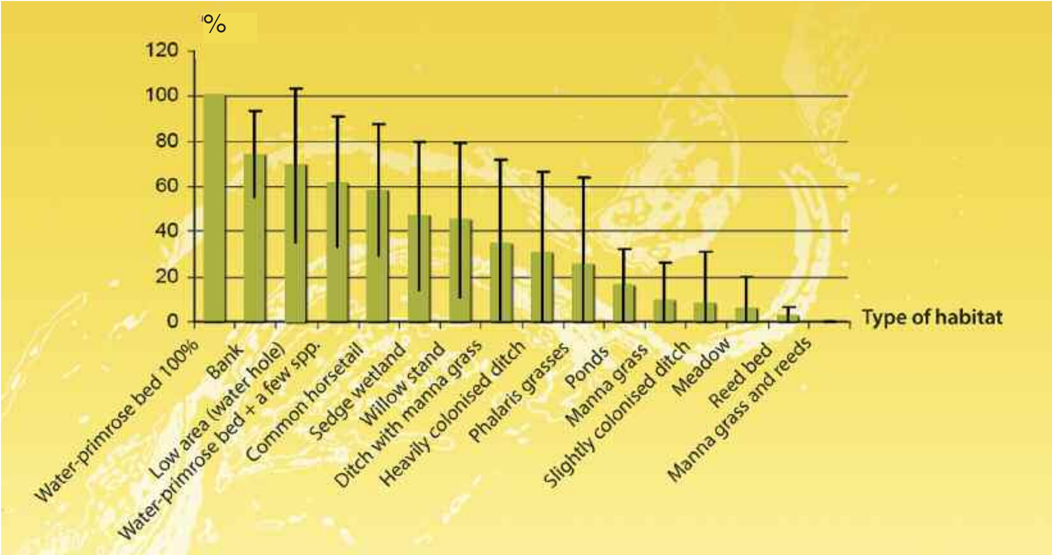
This multiplicity of assessment methods is not in itself a problem. The important aspect is the precision of the data-acquisition protocol and that the observations and/or measurements be systematically carried out in the same manner to enable comparisons.

Assessment of the ecological impacts

For plants, the most common monitoring technique is that of phytosociological surveys carried out before and after management work, thus making it possible to precisely monitor changes in native plant populations.

In the work by Haury *et al.* (2009) on the Gannedel marshes, densities of water primrose varied significantly between the different plant formations sampled, with high densities in low areas and certain ditches, and no plants in areas where common reeds (*Phragmites australis*) and great manna grass (*Glyceria maxima*) were prevalent. This variability in the densities of water primrose appeared to be related to densities of the other species, which were also highly variable, ranging from 0% (a bed comprising only water primrose) to total (or almost total) cover by large helophytes (see Figure 101). The average specific richness of native species in the quadrants was also highly variable, with up to a maximum of almost six species in the quadrants colonised by large helophytes.

Figure 101



Colonisation by water primrose as a function of the type of habitat in the Gannedel marshes. The figure shows the averages and standard deviations of plant cover in 20 quadrants (0.25 square metre) per habitat. According to Haury et al.*, 2009.*

Analysis of the cover by each species in the 324 quadrants studied, representing a total of 55 taxa, not including the water primrose, revealed the continued colonisation by the species primarily in ponds comprising lilies, its low penetration in areas with high levels of manna grass and phalaris grass (relatively tall and dense plant formations), and its adverse effect on species richness and cover by other macrophytes.

More general and consequently less precise assessments can also provide information on ecological processes modified by invasive species. Monitoring carried out on the Turc Pond (Landes department) before and after mechanical uprooting of very large beds of large-flower water primrose present in the shallow pond for over a decade provided information on the modifications caused by the removal and the return of the water primrose (Dutartre, 2004).

Only a small number of native hydrophytes, rigid hornwort (*Ceratophyllum demersum*), two species of pondweed (*Potamogeton crispus* and *P. lucens*) and the yellow water lily (*Nuphar lutea*), were present in the 8-hectare pond prior to the work. The work had a major impact on these species, but they rapidly recolonised the area. Other hydrophytes were observed in the following years, both native species such as large waternymph (*Najas major*) and Eurasian watermilfoil (*Myriophyllum spicatum*), and alien species such as curly waterweed (*Lagarosiphon major*).

Due to a lack of regular maintenance in the pond, water primrose recolonised in less than five years the riparian biotopes where the hydrophytes had appeared, pushing them back once again. The competition for light was easily won by the amphibious alien species. The regular manual maintenance that was subsequently carried out eliminated the water primrose from the pond and facilitated the strong development of the large waternymph.

For animals, the monitoring protocols to assess the ecological impacts following interventions depend on the invasive alien animal species in question and on the available information concerning the native plant and animal species that the alien species consumes or hunts. Counts or population-abundance measurements are occasionally carried out during the year following an intervention, however no standard protocols exist.

One of the difficulties in setting up monitoring of ecological impacts is often the lack of data on the site prior to the intervention and the lack of nearby control sites not yet colonised by the species that could serve for comparisons. In addition, as was shown by the examples mentioned, to our knowledge there are no studies addressing the impacts of IAS management on all the living communities, both fauna and flora. A fairly large amount of information is available on the invertebrate communities colonising aquatic macrophytes, including invasive species, but that information has not been consolidated. Some data may show, for example, the impact of the strong growth of certain invasive plants on the living communities, but no studies are available on the modifications caused in the same environment by the removal of the invasive species.

The difficulty in setting up long-term monitoring programmes, often a result of the lack of long-term funding, is another major handicap in conducting assessments. As the funding needs have risen in step with the increase in the need for management of invasive species, efforts have been made to set priorities for interventions targeting the most troublesome species. The main difficulty in setting priorities lies in selecting the criteria. It is relatively easy to assess the perceived disturbances and, over a fairly large area, to target the species most frequently causing them, however that cannot be the sole type of criterion.

The available knowledge on a species just arriving in an area and that is known to have demonstrated strong invasive capabilities in other parts of the world should result in that species being placed on the list of priorities for intervention, if possible taking into account the data on the potential management techniques and on the impacts of those management techniques. For example, the management techniques for New Zealand pigmyweed are manual uprooting (not always easy) and scraping/stripping of the topsoil or sediment colonised by the species, which also removes the seed banks and the invertebrates from the soil or sediment. Such adverse side effects should be a factor in this species being declared a priority in order to intervene as soon as possible, thus limiting the number of sites requiring work (see the management project in volume 2, page 47) and the corresponding impacts.

The launch of studies to enhance our knowledge on both the ecological impacts of IASs and the impacts caused by their management are indispensable in view of improving our collective capacity to handle the difficulties encountered while minimising the adverse side effects. A further component in this wide-ranging analysis should be the work already being done on assessing the ecosystem services of aquatic environments (Amigues et Chevassus-au-Louis, 2011).

Outlook for improved management of invasive alien species in aquatic environments

IAS management is necessarily based on knowledge and various scientific and technical fields that are all contributing factors to human activities and discussions.

In this context, any possible improvements will be diverse and will probably result from the development of:

- stronger, more effective regulations (see Chapter 2);
- better dissemination and circulation of information (see Chapter 3);
- enhanced knowledge on species biology and management techniques (this chapter);
- monitoring and early-detection networks for IASs (see Chapter 4);
- programmes for applied research (see Chapter 1);
- strategic networks on the various management levels (see Chapters 3 and 6).

In the sections below, two aspects requiring major improvements are presented, namely biosecurity and cost analyses of management interventions in aquatic environments.

Improving biosecurity

This section was drawn and adapted from a report in the first News Bulletin of the Biological invasions in aquatic environments group (see http://www.gt-ibma.eu/activites-du-gt-ibma/lettre-dinformation/lesdossiers-de-la-lettre-dinformation/).

All persons using or simply visiting aquatic environments can, unknowingly, become vectors of pathogens and invasive alien species. Unfortunately, our knowledge on the risks involved in dispersal is totally insufficient. For these reasons, it is important that biosecurity issues be addressed as fully as possible in order to gain new knowledge that can be widely disseminated. Greater awareness of the risks of dispersal and implementation of suitable biosecurity rules could limit the geographic dissemination as well as the disturbances and damage caused by a number of easily transportable invasive species.

Three recent examples of research on this subject will demonstrate the value of this work.

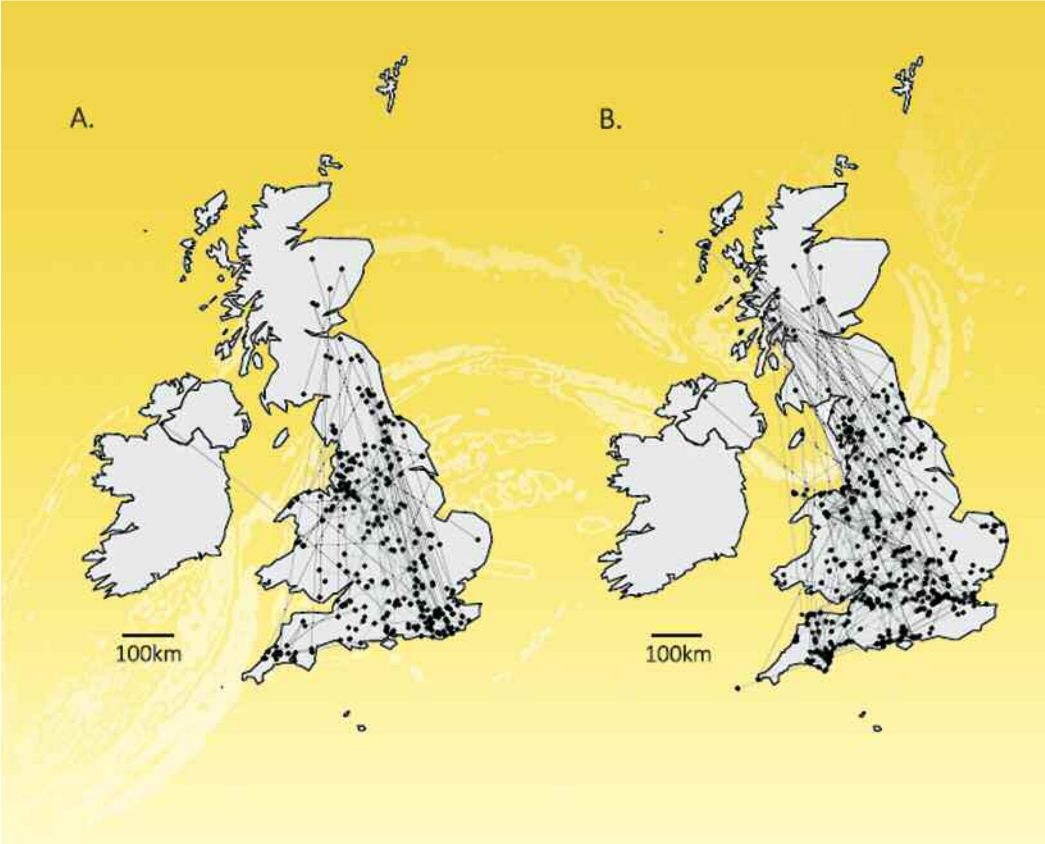
■ Survey of fishing and boating associations in the U.K.

The purpose of the internet survey carried out by Anderson *et al.* (2014) on all British fishing and canoeing/kayaking associations was to study the practices of these people active in aquatic environments to determine the impacts on the dispersal of nine pathogens and ten IASs (plants and animals previously identified as clearly invasive). The questions on practices addressed the cleaning and drying of equipment after use, travel (distances, frequencies and destinations), the number of river basins concerned by travel over short periods, etc. Anglers were also questioned concerning their use and disposal of live bait. The responses to the survey were then analysed in terms of the dispersal risks, ranging from 1 (low risk) to 5 (high risk).

One of the interesting results of the survey is that a majority of anglers (64%) and of boaters (78,5%) used their equipment in more than one river basin in a given two-week period, the time that several of the pathogens and IASs studied could potentially survive. In addition, 12% of the anglers and 50% of the boaters did not clean or dry their equipment between two uses. What is more, almost half of the anglers and boaters used their equipment abroad, primarily in European countries, though in this case only a small percentage did not clean or dry their equipment.

Maps drafted on the basis of the information collected indicate the sites visited by anglers and boaters that travelled to more than one river basin over a two-week period without cleaning or drying their equipment between uses. On the maps, lines linking the sites visited by a person over the two-week period in question clearly illustrate the multiple interconnections that exist between river basins (see Figure 102).

Figure 102



Sites visited by anglers (A) and boaters (B) travelling to more than one river basin over a two-week period without cleaning or drying their equipment between uses. The lines indicate travel between sites. According to Anderson et al., 2014.

Given that over one-third of species introductions in Europe are caused by fishing, recreational boating and other recreational activities, the authors concluded that these uses of aquatic environments risk becoming serious vectors of pathogens and IASs and highlighted the importance of improving the biosecurity aspects of those activities and of raising the awareness of the general public.

If it is acknowledged that the equipment used for these activities can serve as vectors for pathogens and IASs if used without the necessary precautions, what methods could be used to reduce these risks? For example, what cleaning products could be used to eliminate any organisms from the equipment once it is removed from the water?

■ Biosecurity measures to reduce secondary propagation of Asian clams

The work by Barbour *et al.* (2013) on tests of disinfectants for Asian clams (*Corbicula fluminea*) in Ireland produced very useful initial results. The Asian clam, *Corbicula fluminea* (Müller, 1774), is one of the most widely spread invasive bivalves in fresh waters worldwide. The species was first observed in Ireland in 2010. Its rapid spread in the Shannon River confirmed its high colonisation capabilities. The risks of secondary dispersal, caused by human activities, to aquatic environments subject to intense fishing and boating pressures such as the Shannon River were deemed to be high. The objective of the study was to test the effectiveness of methods to remove Asian clams from fishing equipment (nets, waders, other equipment used in fresh water).

The tested products were salt, bleach and a product, Virkon®, specifically developed to disinfect aquaculture equipment (see Figure 103).

Figure 103

© Inland Fisheries Ireland

Disinfection of fishing equipment using Virkon®.

The tests revealed that Virkon® was the most effective product in terms of biosecurity and that it resulted in mortality rates of over 90% for Corbicula fluminea after very short exposure times. According to the authors, to obtain 100% mortality rates, further research would be required concerning both the species biology (stimuli tripping opening of the animal's valves) and other chemical products or combinations of products that could develop synergistic effects.

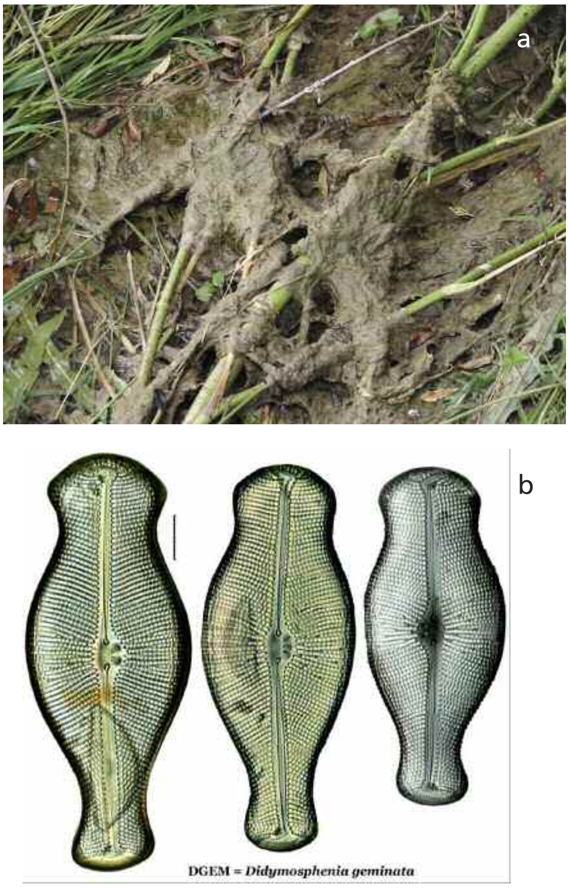
Greater awareness of our potential responsibility in the dispersal of troublesome species may result in biosecurity directives that are seen as further constraints on the use of aquatic environments.

■ The Dydimosphenia geminata diatom

A well-known example of dispersal of an alien species due to certain activities is the *Dydimosphenia geminata* diatom (see Figure 104). This fairly large diatom produces viscous stalks that can attach to sediment and plants. This species can completely cover river bottoms. It originated in the high latitudes and mountainous regions of the northern hemisphere, but has dispersed widely since middle of the 1980s, notably to New Zealand, where it is considered "undesirable", as well as in North America and Europe. Its proliferation impacts the living communities in the colonised rivers. Fishing has been considerably affected in certain areas and clumps of its stalks can block water intakes.

The stalks can attach to the equipment of anglers, boaters and other river users, which explains the great spread of the species. A great deal of information on the species has been disseminated over the past few years in an attempt to draw the attention of people to the dispersal risks and inform on how to avoid them. For example, a document published by the Sustainable development, Ecology and parks, and Natural resources and fauna ministries in Québec in 2008 (www.mddelcc.gouv.qc.ca/eau/eco_aqua/didymo/didymo.pdf), presented the recommendations drafted on the basis of the methods developed and tested by the authorities in New Zealand (*Biosecurity New Zealand*). They deal with how to examine a boat and equipment in view of removing algae before leaving a river and how to clean and dry objects that were in contact with the water, stressing notably how to deal with absorbent materials such as the felt pads beneath the boots of anglers.

Figure 104



a © A.Dutartre, Irstea
b © Coste et al.

The Dydimosphenia geminata *diatom attaches to sediment and plants. It has spread widely via the equipment used for recreational activities.*

In the field of biosecurity, we are confronted with our insufficient information (that must be improved) and with the need to modify certain practices in aquatic environments (here the public must be convinced). These changes will necessarily take time, but they would appear indispensable if we are to improve IAS management.

Better understanding the costs of IAS management and the comparative economic analyses of the potential methods

One of the constant difficulties encountered by management projects is their justification in economic terms. The high and steadily rising costs of IAS management on all levels, national, European and worldwide, are increasingly seen as difficult to accept and, particularly in Europe, as directly competing with the funding needs for the restoration of aquatic environments undertaken in compliance with the Water framework directive.

Most of the available economic studies assess the damage caused by IASs and their management costs in consolidated terms, combining all species, which is of course necessary, but not sufficient. The world is confronted with multiple biological invasions, each causing specific damages and requiring different management techniques. They can be analysed from the financial standpoint, but studies should distinguish between the different types of IAS in order to highlight the issues of each management situation.

The dispersion of the invasive species to be managed in environments is one of the difficulties involved in correctly determining the economic costs. Concerning aquatic plants, the management technique selected is not the only parameter. For example, two inventions may use the same equipment, but for the first, the plant beds are dispersed, resulting in "downtime" travel between the beds (see Figure 105), whereas for the second, the equipment works continuously on a single, dense bed (see Figure 106). The average cost per unit volume of extracted plants or per unit of treated surface area is directly related to the dispersion of the plants, to the distances travelled by the equipment, to the conditions for access by the equipment and for unloading/loading, etc.

Figure 105

© A.Dutartre, Irstea

Dispersed beds of large-flowered waterweed in the Pen-Mur reservoir (Morbihan department).

Figure 106

© A.Dutartre, Irstea

Dense beds of curly waterweed in Blanc Pond (Landes department).

As part of the Water-primrose project in the INVABIO programme (Dutartre et al., 2007), management costs for water primrose were analysed in order to determine the best management conditions in economic terms (Million, 2004). The main results expected from the analysis dealt with the intervention frequencies and time periods as a function of the type of environment, and with the regular management of the water primrose remaining on the sites. The study could not answer all the questions raised, however it did provide a rough estimate of the costs for the two main techniques (manual and mechanised uprooting) used to regulate water primrose.

The average cost per ton of fresh biomass of uprooted water primrose ranged from 1 100 to 1 330 euros for manual work and from 51 to 64 euros for mechanised work.

One of the conclusions of the study was that the solution for water-primrose management that seemed to produce the best results was an intervention severely reducing water primrose on a site followed by regular work on the remaining plants, i.e. precisely the solution already adopted on a number of sites such as the Marais Poitevin marshes.

The analysis of intervention costs for invasive aquatic plants in the Pays-de-la-Loire region conducted by Matrat *et al.* in 2011 (http://www.pays-de-la-loire.developpement-durable.gouv.fr/2011-colloque-regional-les-plantes-a1338.html) used the data from 317 of the 449 sites listed since 1994, thanks to the documents filed by managers prior to 2006 and the site-report sheets used since. The total cost of the interventions amounted to approximately 3.5 million euros. This analysis produced a number of data points including average costs for plant uprooting, all species combined, as a function of the surface area treated and of the volume extracted. The average costs per square metre ranged from 0.40 euros for surface areas greater than one hectare to approximately 35 euros for surface areas of less than ten square metres. The average costs ranged from 4.20 euros for volumes greater than 100 cubic metres to approximately 2 300 euros for volumes of less than 0.1 cubic metres.

These results for costs per unit of surface area or per unit volume/biomass of extracted plants are very useful for future studies on management techniques, but they provide very little information on the issues surrounding interventions. They do not reflect the specificities and ecological value of the environments managed, nor the characteristics of the targeted invasive species and the site, e.g. dispersion of the plant beds, access conditions for equipment, etc.

The management costs for water primrose paid by IIBSN in the Marais Poitevin wet marshes exceed 200 000 euros per year (Pipet and Dutartre, 2014). That is a great deal of money, however a comparison between the management costs for water primrose and the cost of “doing nothing” puts the management costs into perspective.

An economic assessment of the theoretical costs of the damage if the entire hydrographic network of the wet marshes were to be colonised by water primrose was carried out by Aline Issanchou in 2012. She included in the calculations the human uses of the site, notably tourism and boating in the Marais Poitevin marshes, the risks in terms of flooding and other values, e.g. recreational and aesthetic values. For example, she estimated that annual tourism revenues in the Marais Poitevin marshes amounted to almost 145 million euros, a figure that puts the annual management costs for water primrose into perspective given that the attractiveness of the marshes is due in great part to boat rides in the “Green Venice” that would become impossible if the channels were invaded by water primrose. Though this analysis requires more work due to the lack of precise data concerning certain elements, the author nonetheless concluded that according to the assessment method implemented, the costs of the damage caused by water primrose, *“not all of which were taken into account in the analysis”* would appear to be far greater than the management costs effectively incurred. According to her calculations, *“starting at 200 tons of fresh, water-primrose biomass, the total damage is estimated at 82 million euros”* (Pipet and Dutartre, 2014).

The Sustainable-development division of the Ecology ministry is currently conducting a study on the national level to ascertain the overall costs of the damages caused by IASs on the environment, for human health and the economy, and the costs of management interventions. Following an initial review of the literature, a questionnaire was sent to the main stakeholders in the beginning of the summer of 2014. The study is an element in the implementation of the new European regulation. In addition to the assessable economic losses, data on *"harm to the well-being of stakeholders and on any non-market benefits (recreational activities, aesthetics, amenities, etc.)"* will be included in the analysis.

These economic aspects do not include the ecological impacts of interventions nor the ecosystem services provided by environments. Invasive plants can occupy the same biotopes as important, native plants (see Figure 107). Which criteria should be used to decide whether to intervene? Should the entire site be cleared, knowing that bogbean is not very competitive and will probably not recolonise the site, or should the intervention target just the two invasive species, knowing that the cost of the intervention will be much higher?

With manual uprooting, only the invasive plants are removed and the native species remain (see Figure 108), even when the surface areas are very small. For example, a trained operator can differentiate between large-flower water primrose (*Ludwigia grandiflora*), with alternating leaves, and Marsh seedbox (*Ludwigia palustris*), a native species with a red stalk and opposing leaves.

Continued work on these analyses, on the national or regional levels, or species by species, would appear necessary to improve IAS management strategies. This work would be of assistance in defining a comprehensive strategy driven on the national level, in compliance with the European regulation, as well as more specific regional and local strategies. But in all cases, this work should be accompanied by an assessment of the ecosystem services provided by aquatic environments (Amigues and Chevassus-au-Louis, 2011) to ensure that all aspects of the ecological, social and economic issues involved in the management of invasive alien species are fully taken into account.

Figure 107

© A.Dutartre, Irstea

In the foreground is a stand of yellow iris (Iris pseudacorus) surrounded by a small bed of bogbean (Menyanthes trifoliata), a species that is slowly disappearing from lakes along the Aquitaine coast. Just behind is a dense bed of parrot-feather watermilfoil with some water primrose. Léon Pond (Landes department).

Figure 108

© A.Dutartre, Irstea

Manual uprooting of water primrose in a ditch linked to the Noir Pond (Landes department).