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Hiroyuki Matsuda *Editor*

Ecological Risk Management

For Conservation Biology and Ecotoxicology

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Cover illustration: The brown bear and its cub at Rusha River. (Photo taken at Rusha River in Shiretoko, Hokkaido, Japan on July 9, 2004, by Dr. Koichi Kaji)

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Contents

Chapter 1
Introduction: What Is Risk Science?

Hiroyuki Matsuda

Abstract Environmental risks can be broadly divided into human health risks and ecosystem risks. This book contains many case studies on ecological risk management in Japan. There are various ways of thinking about environmental risks. For example, Japan develops science later than the West, which often mimics Western science policy, but it also has some unique aspects about the precautionary principle and risk management. In this chapter, we will explain environmental risks and ecological risks from a general risk concept, using some examples of ecological risks described throughout this book. Risk is characterized by endpoint or undesired event, hazard when it happens, and the probability that it happens. Therefore, risk assessment uses probability theory and statistics. In addition, risk assessment is often based on unverified assumptions. In that sense, risk science is beyond normal science. Understanding the difference between the two scopes will be an introduction to the precautionary principle and risk science. In this book, I will introduce various measures against ecological risks based on precautionary measures. The differences in the scope of the precautionary measures may differ between decision-makers, or between the ecological and health sectors.

Keywords Precautionary principle · Mercury poisoning · Unverified assumption · Risk assessment · Human health risk · Ecological risk

1.1 What Is Risk?

Human impact on the environment is too large and broad to recover, but the magnitude of the impacts is still unclear. The **precautionary principle**, Principle 15 of the Rio Declaration "In order to protect the environment, the precautionary approach shall be widely applied by States according to their capabilities. Where there are threats of serious or irreversible damage, lack of full scientific certainty

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shall not be used as a reason for postponing cost-effective measures to prevent environmental degradation," was agreed in United Nations Conference on Environment and Development (UNCED) in 1992. At least after Rio Declaration, we are to prevent risks without full scientific evidence. It makes a scientific field of "risk science" and "regulatory science" (see Chap. 16).

Also in ecology, a precautionary principle was agreed in 1992. In an article of Convention on Biological Diversity, "[n]oting also that where there is a threat of significant reduction or loss of biological diversity, lack of full scientific certainty should not be used as a reason for postponing measures to avoid or minimize such a threat."

Risk is usually defined as the product of hazard and its possibility of occurrence. To define risk, we should clarify an **endpoint**, which is an undesired event. In the context of human health risk, death of a patient is one of the typical endpoints. In the context of business risk, bankrupt of a company is one of the endpoints. In the context of ecological risk, extinction of a species is one of the endpoints.

If an endpoint has once been defined, we would evaluate the magnitude of **hazard** when an endpoint occurs. In human health risk, hazard of a person's death is usually equal because we should avoid discrimination of human being between rich and poor, male and female, and young and old persons, and between races. Unlike human death, there is a difference in hazard between bankrupts of large and small companies in business risk. There may also be a difference in the magnitude of hazard of species extinction. For example, extinction of a large mammal species may be recognized as a larger hazard than that of a small insect species. In addition, the magnitude of hazard of species extinction is probably uncertain.

Assessment of the probability that an endpoint happens is often difficult and uncertain. There is often a lack of full scientific certainty in risk assessment. In population ecology, the survival rate and reproductive rate of organisms are considered for the mathematical model describing conditions of the populations. Uncertainties and environmental variations when measuring are considered for the population size and other indicators. As a result, future prediction can only be made in a probabilistic way. Risk evaluation to show an interval estimation is required. If management measures are predetermined, it is impossible to cope with a contingency. As a management method to cope with such uncertainty, I recommend **adaptive management** (AM). AM is to make a management plan based on unproven conditions, continuously monitor changes in state while implementing the management and review the measures as required, and verify the adequateness of the conditions (Walters 1997). It is important to predetermine the method of reviewing the measures and the method of verifying the conditions.

In this chapter, I introduce a difference in thinking before and after the adaptive risk management is established, describing three cases: fisheries management (Chap. 5), wildlife management (Chaps. 9, 11, 12, 13), and environmental impact assessment (EIA, Chaps. 8, 10).

1.2 What Is Ecological Risk?

Human health risk and ecological risk are two major risks in environmental risk science. The endpoint of human health risk is usually personal death, getting cancer, or other "fatal" events of persons. The endpoint of ecological risk is often related to loss of biodiversity, in a typical way, extinction of some **species** or extinction of some local populations. More generally, the endpoint of ecological risk is related to loss of ecosystem services or loss of nature's contribution to people (Díaz et al. 2015).

The word before "risk" expresses either type of endpoint or risk factors. For example. "extinction risk," "cancer risk," and "risk of management failure" express endpoints of these risks. On the other hand, "radiation risk" and "GMO (genetically modified organisms) risk" express risk factors. Do not confuse them.

Species extinction is definitely irreversible damage to biodiversity. Species extinction is somewhat natural phenomenon partly because of geological climate change. In fossil records, the average "longevity" of species is 1–10 million years, or the rate of species extinction is 0.01–0.1% per thousand years (Fig. 1.1). Recently, at least after the twentieth century, it has increased by 100–1000 fold. In addition, the rate will increase by 10–100 fold in the future, mainly due to the impacts of human activities. Note that IUCN's estimation is based on a precautionary approach. Even though it might result in overestimation, the extinction rate is much larger than it was. Some of the species extinction made a significant change in ecosystem structure, and probably ecosystem functions. In Japan, after wolves went extinct by 1890 (Kaji et al. 2010), deer has been enemy-free and its population increased. Extinction of top predators definitely changed ecosystem structure.

Biodiversity or biological diversity means the variety of living things, including rare and common species, used and unused species. Biodiversity also means the variety of ecosystems, the abundance of different species, and the genetic variety within a species living within a particular region.

For local ecosystems and local societies, extinction of the local population may be a significant damage on local ecosystem functions. In biology, **population** means a set of organisms of a particular species as an inter-breeding unit of more or less isolated. Population also means the number of individuals in the population. In order to distinguish from population in the sense of demography and statistics, it is also called biological population.

An ecosystem is a dynamic complex of plant, animal, and microorganism communities and the nonliving environment interacting as a functional unit. An ecosystem is either primeval or modified by human use. **Ecosystem services** are the benefits people obtain from ecosystems. There are four types of ecosystem services. Provisioning services are fiber, food, water, timber, and biofuels. Regulating services affect climate, floods, disease, wastes, and water quality. Cultural services provide recreational, aesthetic, and spiritual benefits. Supporting services are soil formation, photosynthesis, and nutrient cycling (Millennium Ecosystem Assessment 2005). The human fundamentally depends on the flow of ecosystem services.

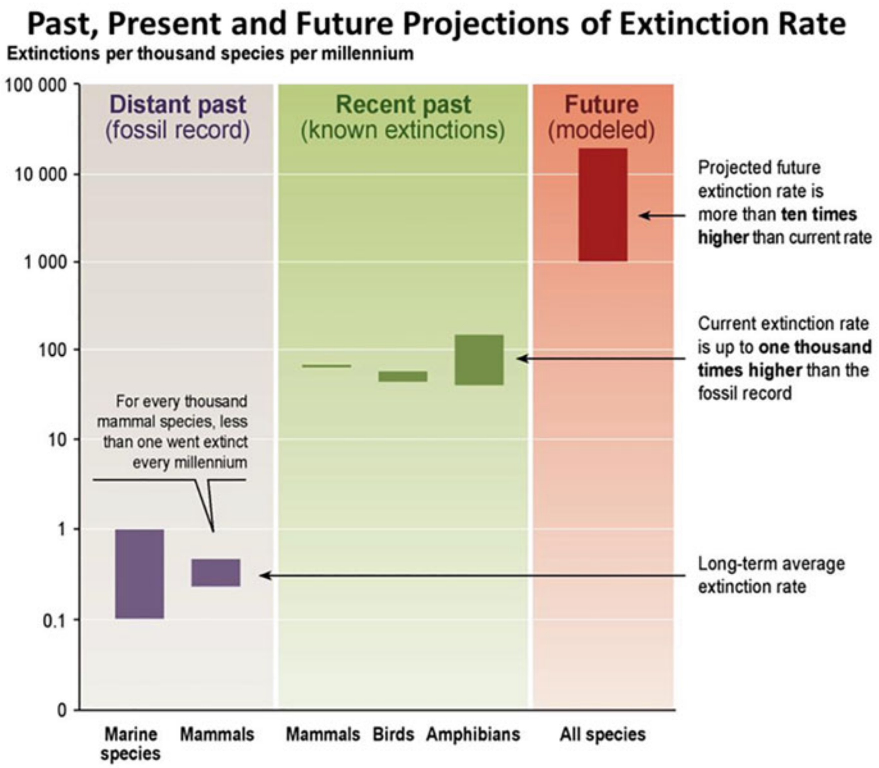

Fig. 1.1 Species extinction rates in the fossil, recent and future projection (Millennium Ecosystem Assessment 2005) are derived from a variety of different models, all based on current trends, but considerably uncertain (as indicated by the wider range)

There are other types of endpoints in ecological risks. In population management, the endpoint is the failure of management. In fishery management of southern bluefin tuna (SBT, *Thunnus maccoyii*), Commission for the Conservation of Southern Bluefin Tuna (CCSBT) has the target of recovering the SBT to the 1980 level by 2020 (Mori et al. 2001). In brown bear management in Japan, the purpose of management is to ensure its viability and reduce the number of intrusions into crop fields to an acceptable level (Ohta et al. 2012). For that purpose, two endpoints are defined; the bear population being below 25% of the estimated population size in 2008 and the number of nuisance female bears exceeding the average maximum estimated number of nuisance females between 2001 and 2003. In environmental impact assessment, endpoints are some predictions described in the environmental impact statement, e.g., survivorship of transplanted endangered plants, being false (Matsuda et al. 2003).

Ecological risk assessment has first been developed in ecotoxicology. In analogy to human health risk due to environmental chemicals, bioassay plays an important role in ecotoxicology. In the decision of Japanese environmental standard of zinc

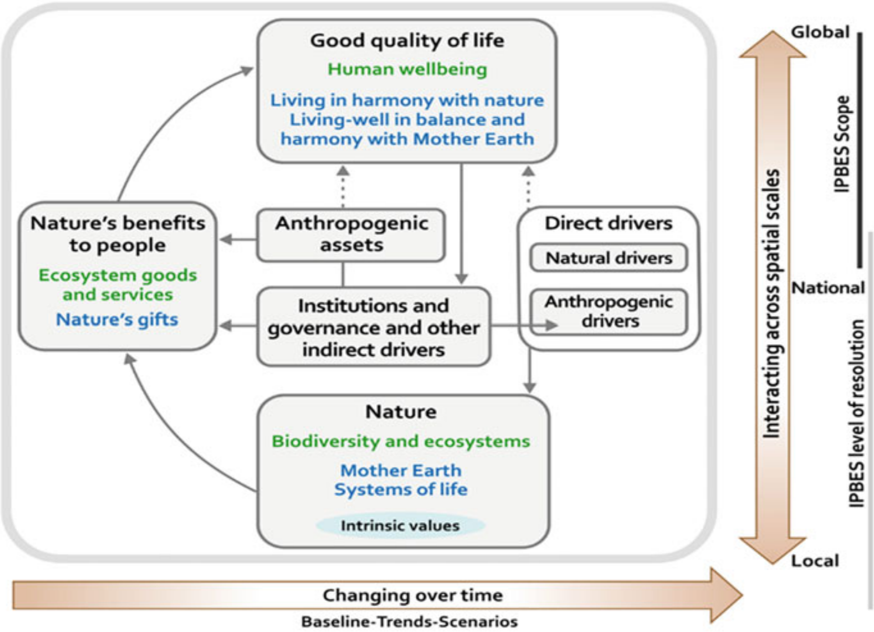

Fig. 1.2 Conceptual framework between human activities, ecosystem services, and human well-being (Díaz et al. 2015)

concentration in freshwater ecosystems, single-species toxicity tests in the laboratory usually estimate toxicity thresholds for individual organisms. The endpoints of ecological risk of zinc are toxicity in survival, growth, or reproductive measures of insects. The Japanese standard of zinc concentration is determined by chronic toxicity to a macroinvertebrate species (Chap. 4). However, this value may be too conservative for the prevention of population extinctions because it is not based on population-level risk for any species. It is necessary to investigate macroinvertebrate populations in rivers (Kamo and Naito 2008).

Millennium Ecosystem Assessment (2005) focuses on the linkages between ecosystems and human well-being and ecosystem services. Figure 1.2 describes the conceptual framework between human activities, ecosystem services, and human well-being. Human impacts affect ecosystem services through indirect and direct driving forces. Such forces change in ecosystems and thereby causing changes in human well-being. At the same time, social, economic, and cultural factors unrelated to ecosystems alter the human condition with natural forces.

Human well-being includes many aspects of our daily life. It includes material well-being, relationships with family and friends, emotional and physical health, and freedom of choice. Convention on Biological Diversity uses this term as an ultimate reason of biodiversity conservation (Díaz et al. 2015). Although this is utilitarianism, international society may not ignore any idea beyond utilitarianism. Convention on Biological Diversity describes “Ecosystems should be managed for their intrinsic values and for the tangible or intangible benefits for humans, in a fair and equitable way" in Principle 1 and "Biological diversity is critical both for its intrinsic value and because of the key role it plays in providing the ecosystem and other services upon which we all ultimately depend" in Principle 11. I usually use the advertisement of Mastercard® as an analogy to limit utilitarianism: "There are some things money can't buy. For everything else, there's MasterCard." If conservation of ecosystem services that support human well-being is a reason why conservation of nature for consensus of international convention, it does not imply the protection of "priceless" values of nature.

Direct driving forces affect biodiversity and ecosystem services. For example, overexploitation of fish reduces future stock that is one of the provisioning services, and it may reduce food security that is one of the human well-being. A reason for overexploitation may be overpopulation or globalization in the economy, which are indirect driving forces. In addition, change in human well-being may change indirect driving forces in the future.

Five main pressures affect biodiversity and ecosystem services: land use change in habitats, overexploitation of natural resources, climate change, invasive alien species, and pollution, according to "Global Biodiversity Outlook 3" (Secretariat of the Convention on Biological Diversity 2010). In the National Biodiversity Strategy of Japan, three main crises on biodiversity are detected; (1) species and habitat degradation due to excessive human activities, (2) degradation of satochi-satoyama due to insufficient level of management, and (3) ecosystem disturbances caused by the introduced alien species and chemical contaminations (The Government of Japan 2010). Satochi and satoyama mean plain and hilly landscapes formed by sustainable use of natural resources, respectively. Crisis 1 corresponds with land use change and overexploitation. Crisis 3 corresponds with invasive alien species and pollution. Crisis 2 is not described in the five driving forces of Millennium Ecosystem Assessment (2005). The National Biodiversity Strategy of Japan recognizes that human impacts are not always negative, sustainable use at least sometimes enhances biodiversity in comparison with it under no human impact.

1.3 Three Types of Risks

Majority of **risk assessment** method is common between business risk and environmental risk. We define endpoint as an event that should be avoided, hazard as the magnitude of impact when an endpoint happens, probability that endpoint happens. Risk is the product of hazard and the probability. Especially in environmental risk assessment, we evaluate the probability, and sometimes the magnitude of hazard, by unverified assumptions, sometimes even though we realize unrealistic assumptions that we use. In the context of toxicology, these unverified assumptions or hypotheses are called "**scenario.**" Typically, an exposure scenario means the set of conditions, including operational conditions and risk management measures, that describe how the substance is manufactured or used during its life-cycle and how the manufacturer

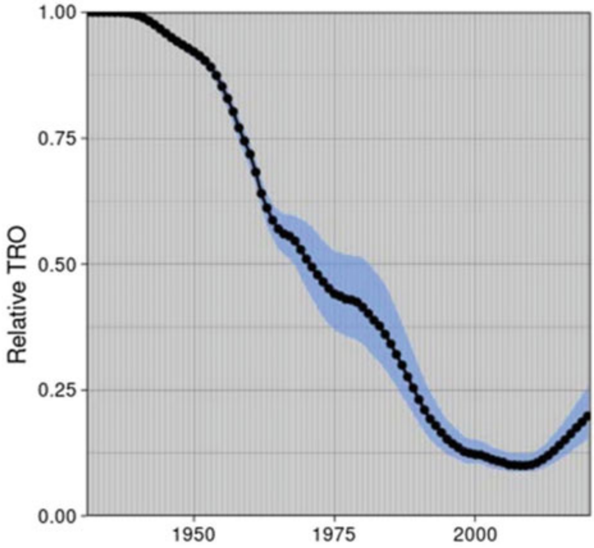

Fig. 1.3 Trends in relative total reproductive output (TRO, which is a proxy of the spawning stock abundance) from 1931 to 2020 estimated for the current reference set of the operating models. Black dots and blue area show median and 5–95 percentiles, respectively (CCSBT 2017)

or importer controls, or recommends downstream users to control, exposures of humans and the environment. I note that the scenario also depends on policy or management plan.

In the case of extinction risk assessment for southern bluefin tuna (SBT), we have data of past trends of population size from 1969 to 1997 (Fig. 1.3). SBT was listed as Critically Endangered by The International Union for Conservation of Nature (IUCN) in 1996 because the population decline rate during the past three generations satisfies Criterion A of IUCN's Redlist Criteria (see Chap. 8). Using the data until 1995, the population decline rate of SBT is >80% within the past 25 years. The extinction risk within the next 70 years, using the assumption that the past population decline rate, the average and its SD, will continue in the future (see Chap. 6).

CCSBT (Convention on Conservation of Southern Bluefin Tuna) agreed on the regulation of catch quota in 1989 to recover the population to the 1980 stock level by 2020. Therefore, the assumption that the past population decline rate continues in the future is probably unrealistic. On the other hand, the management may not always succeed. In fact, the SBT stock did not recover from 2000 to 2010 as was planned. CCSBT gave up the above numerical goal in 2003 and agreed with a revised numerical goal in 2011. Anyway, we might consider another assumption that the population decline rate will decrease after the management plan was agreed. Risk scientists usually adopt a simple assumption such as a linear relationship. If nonlinear relationship is adopted, nobody knows the degree of nonlinearity. It is controversial between scientists and other stakeholders. There is little room for arbitrary operation with simple assumptions. When there is a lack of full scientific certainty, we must adopt either of the multiple hypotheses because we do not know which is true. The magnitude of risk definitely depends on which assumption is adopted. Risk assessment document must describe which scenario is used. In addition, which scenario is adopted should not be determined only by scientists, but by stakeholders. We usually choose the simpler and precautionary scenario.

We are not satisfied solely with risk assessment. Our purpose is to control the risk on a socially allowable level (Chap. 17). This is called **risk management**. Risk management consists of the process of identifying, quantifying, and managing the risks. Risks include strategic, operational failures, unforced errors, environmental disasters, and regulatory violations. While it is impossible that managers remove all risk, it is important that they properly understand and manage the risks that stakeholders are willing to accept in the context of the overall strategy. The manager is primarily responsible for risk management, but the board of scientists, internal auditor, external auditor, and general counsel also play critical roles.

Our purpose is not solely to decrease the risk of a particular endpoint. Seeking "zero" risk is often too expensive. In Principle 15 of Rio Declaration, "lack of full scientific certainty shall not be used as a reason for postponing cost-effective measures to prevent environmental degradation." Therefore, we need to consider cost-effectiveness of risk management measures. **Risk-benefit analysis** is as important as cost-benefit analysis. Unlike cost-benefit analysis, the dimension of risk often differs from that of benefit (money).

In addition, there are often tradeoff relationships between different kinds of risks. For example, the construction of wind farms contributes mitigation of climate change, but it also increases avian collisions of endangered raptors. This situation is called **risk tradeoff**.

There are three kinds of uncertainties or errors in risk management. First, we do not know the exact information about socio-ecological status, e.g., the stock size of SBT. This is called **measurement errors** or observation uncertainties. Second, the environmental condition temporally changes. e.g., the per capita recruitment of tuna changes from year to year depending on environmental conditions such as sea surface temperature. This is called **process errors** or process uncertainties (Hilborn and Mangel 1997). Finally, **operational error** or implementation error is one of the human errors that means malfunctions, accidents, or other unintended consequences that differ from human actions that are expected by the management plan. In the case of southern blue fin tuna fishery, the true catch amount was more than the reported catch from 1996 to 2005 (CCSBT 2017). In deer management on Hokkaido Island, the actual catch in number in 2011 was smaller than that determined by the action plan.

As mentioned above, risk management is a process of identifying, quantifying, and managing the risks. Stakeholders play critical roles. **Risk communication** is exchange of information and opinions, and effective dialog among those responsible for the risk management process and those who may be affected by the outcomes of those risks. Stakeholders do not always seek minimizing risks. Mis-communication is usually a critical source of mismanagement, if stakeholders do not know the exact risk level, management cost and opinions of other than themselves, or if their opinion is not sufficiently involved for decision-making, they hesitate to accept some level of risk or they are unconscious of identifying the risk. In other words, risk communication is a human dimension of the risk management process.

Majority of **risk assessment** method is common between business risk and environmental risk. We define endpoint as an event that should be avoided, hazard as the magnitude of impact when an endpoint happens, the probability that endpoint happens. Risk is quantitatively defined as the product of hazard and the probability. Especially in environmental risk assessment, we evaluate the probability, and sometimes the magnitude of hazard, by unverified assumptions, sometimes even though we realize unrealistic assumptions that we use. In the context of toxicology, these unverified assumptions or hypotheses are called "**scenario.**" Typically, an exposure scenario means the set of conditions, including operational conditions and risk management measures, that describe how the substance is manufactured or used during its life-cycle and how the manufacturer or importer controls, or recommends downstream users to control, exposures of humans and the environment. I note that the scenario also depends on policy or management plan.

1.4 Risk Science as a Part of Regulatory Science

Risk science relates to "regulatory science" defined independently by Uchiyama (1987) and Jasanoff (1990). According to Jasanoff (1990), the scientific and technical foundations underlying regulations are in variety of industries, especially those involving health and safety. Regulatory science is characterized by processes of determining safety standards. Safety is defined in the third edition of the International Organization for Standardization (ISO)/International Electrotechnical Commission (IEC) Guide 51(7) as "free from unacceptable risk." "For individuals, safety is the combined concept of objective risk assessment and subjective risk perception, and for society safety is a concept composed of risk assessed in a reproducible manner and social consensus related to the level of risk that is not tolerable" (Murakami 2016). Safety can be judged using regulatory science rather than normal science. "Regulatory science is useful for filling the gaps between pure scientific knowledge and decision-making. . . .Risk analysis is conducted within the framework of regulatory science and allows us to provide information and to update scientific knowledge for decision-making in the face of uncertainties" (Murakami 2016).

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Part I
Risk Assessment Based on Ecotoxicology

Chapter 2
How to Determine the Relief Target for Minamata Disease

Hiroyuki Matsuda and Sin'ya Ueno

Abstract Minamata disease that occurred in southern Kyushu and Niigata prefecture in Japan was caused by ingesting seafood that is contaminated by methylmercury discharged from factories. This is a world-renowned pollution disease. Minamata disease is judged from the symptoms peculiar to mercury poisoning. However, it is known that mercury poisoning causes other disorders even at low concentrations of exposure, especially in fetal exposure. Those who are not certified as Minamata disease patients is widely rescued if they have eaten seafood in the area and have some kind of disability. In this chapter, we will discuss the concept of the precautionary principle and its scope of application through the case of Minamata disease. The key concepts are benchmark dose (BMD) and its lower limit (BMDL). When a person's symptoms occur at a certain frequency, say 5%, without mercury intake, with a higher probability of a certain exposure of mercury intake, say 10%, this exposure level is called BMD. BMD is used for tolerable intake. The relationship between this exposure and the incidence of symptoms is estimated by a dose–response relationship, but the estimation includes uncertainty. Based on the precautionary principle, the fifth percentile of BMD is calculated. This is called BMDL. It is used as a recommendation based on the precautionary principle as a guide to refrain from mercury intake. In Japan, this is used as the standard for people who are eligible for relief from Minamata disease. In that case, 5% of people who are not actually exposed to mercury will be eligible for relief, and there will be less than 10% of true relief recipients due to mercury exposure.

Keywords Precautionary principle · Mercury poisoning · Unverified assumption · Risk assessment · Human health risk · Ecological risk

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2.1 What Is Minamata Disease?

First, we consider human health risk of methylmercury in seafood. This is a good introductory chapter to study environmental risk assessment because a basic methodology of ecological risk management is based on the methodology of human health risk assessment. In addition, seafood safety is important in fisheries science. Japanese people eat a lot of seafood, so they are highly exposed to mercury. The international treaty on mercury control is called the Minamata Convention on Mercury, which entered into force on August 16, 2017.

Food Safety Commission of Japan (FSCJ), released the guideline "Re-examination (outline) of notes about the ingestion and mercury of seafood to pregnant women" on August 12, 2005. About the mercury concentration of seafoods, various criticisms are taken out to the former guideline by FSCJ. I expected the revised guideline that was reflected by these criticisms in this reexamination. For Japanese, seafood is recognized as indispensable to healthy eating habits and has the outstanding nutritional attribute. Since the mercury which exists in nature is accumulated in seafood in process of a food chain, the mercury concentration of some seafood is high as compared with other species. Some scientists reported anxious about the possibility that low-concentration mercury ingestion will affect an embryo. A pregnant woman is considered that fixed cautions are required for ingestion of mercury. It is needed to control the frequency of eating seafood including whales, especially for woman under pregnancy.

Minamata disease is a toxic central nervous system disease caused by methylmercury compounds derived from industrial wastewater. The first case was confirmed in Minamata City, Kumamoto Prefecture in May 1956, which was named "Minamata Disease" in 1957. In Japan, Minamata disease, Niigata Minamata disease in the Agano River basin, Niigata prefecture, Itai-itai disease due to cadmium poisoning, and Yokkaichi asthma in Mie prefecture are called four major pollution diseases

It was attributed to methylmercury emitted by a company in Minamata that produced acetaldehyde, but the cause was so controversial that the identification of the cause was delayed. While experts at the local Kumamoto University School of Medicine claimed the methylmercury theory from the outset, experts from the University of Tokyo disagreed, who were called the "running-dog of the government."

Minamata Bay and the surrounding Shiranui Sea (Yatsushiro Sea) are fishing grounds. Eating polluted seafood has not only affected the health of fishermen but also had a devastating effect on fishing activities. Initially, many fishers' families hidden the health damage and did not apply to Minamata disease (Tsurumi 2006).

The company was initially not responsible but moved the outlet of the water outlet bay to discharge near the mouth of the Minamata river in the Yatsushiro Sea. As a result, polluted areas have expanded, and more people were exposed to methylmercury. In 1973, the "Pollution-Related Health Damage Compensation Law" was enacted. As of the end of February 2016, there were 2280 people, including the deceased, which has been certified by the government for Minamata disease.

The fishery was closed in 1958, and many fishermen and residents applied for certification of Minamata disease. Methylmercury intake in the certified patients at that time was quite high. However, there would have been less exposed residents and fishermen in the vicinity. How far to certify the less exposed people was one issue. At this stage certification who were exposed to high methylmercury intake was medically documented as a disorder. The number of new Minamata disease certified persons has been very small recently. This is probably because the pollution source disappeared around 1969.

On the other hand, the Act on Special Measures Concerning Victims of Minamata Disease and solution to the Problem of Minamata Disease was enacted in 2009. The government has decided to take "as many as possible" in order to rescue residents. Many have been rescued in addition to those who have been certified as Minamata disease.

There are two types of "victims of Minamata disease" in Japan. One is the victims of the Minamata disease certified by the Examination Committee for Minamata Disease (hereinafter referred to as the "certified patients") who have been accredited based on the Pollution-Related Health Damage Compensation Law. The other is a victim recognized under the Minamata Disease Special Measures Law (hereinafter "relief patients"). The condition is that a person has lived in a polluted area designated by the government for a certain period, consumed seafood in the polluted area, and has peripheral limb or systemic sensory disabilities. In 2019, The relief targets were extended to those who lived in mountainous areas surrounding the polluted areas.

Exposure in affected adults and during pregnancies in Minamata was very high, as reflected in maternal hair mercury concentrations that ranged from above 50 mg/kg up to a maximum of 705 mg/kg (Harada 1995). In 1972, the consumption of seed treated with methylmercury fungicide in Iraq resulted in the poisoning of several thousand inhabitants, again with newborns and infants seen as the most vulnerable group for neurotoxic effects" (EFSA 2012).

Minamata disease certification is limited to those with multiple symptoms specific to methylmercury poisoning. There were approximately 3,000 certified patients in Kumamoto, Kagoshima and Niigata prefectures. The number of officially certificated victims has not increased because the government stipulated that victims must exhibit multiple symptoms centering on sensory dysfunction in 1977.

In August 1985, in the Second Minamata Disease Lawsuit, the Fukuoka High Court recognized plaintiff patients who had previously been denied certification by the Government, as being victims of Minamata disease. Chisso did not appeal against this decision and the case was settled.

The Government did not change the criteria for the certification, but they introduced the Special Medical Project in 1986. The project paid the obligatory national health insurance contributions for relief patients who were not certificated for Minamata disease, but had eaten polluted seafood and suffered from sensory disturbances in their arms or legs.

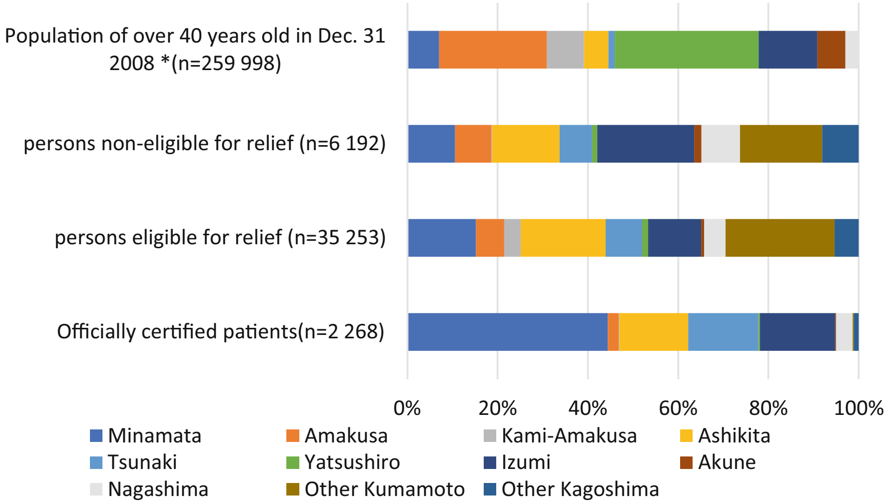

Fig. 2.1 Statistical data on administrative measures related to Minamata disease as of the end of October 2008 (Takaoka 2016). *Other Kumamoto and Other Kagoshima are not included in the population of over 40 years old

By the year 2000, a total of 2263 patients in Kumamoto and Kagoshima Prefecture were recognized as Minamata disease victims, and a further 10,350 had received relief under the Comprehensive Measures of Minamata disease. After that, other trials continued, and the number of relief patients increased, but the government's criteria for certification has not changed and the number of certified patients has hardly increased (Fig. 2.1).

Although the criteria for certification were initially applied to the precautionary principle, the final criteria for certification were limited to those with the unique symptoms of methylmercury poisoning (Yorifuji et al. 2013). In contrast, relief measures are widely adopted in judgments. People eligible for relief may include those who have symptoms that are not caused by methylmercury exposure.

Therefore, I need to evaluate the health risk assessment based on the effects of mercury from factory effluent and the daily intake of mercury from seafood. Seafood safety is one of the most important food safety concerns. In this chapter, I focus on this problem. I will develop the method for designing the risk management of one's eating habits.

2.2 Average Intake of Mercury by the Japanese

EFSA [European Food Safety Authority] (2012) established a tolerable weekly intake (TWI) for methylmercury of 1.3 μg/kg-body weight expressed as mercury. "Exposure to methylmercury above the TWI is of concern, but if measures to reduce methylmercury exposure are considered, the potential beneficial effects of fish consumption should also be taken into account." The JECFA [Joint FAO/WHO Expert Committee on Food Additives] considered the trade-off between the adverse effects of mercury exposure from fish foods and the benefits of ingesting unsaturated fatty acids." "The JECFA selected the BMDL05 [the benchmark dose level with 5% of BMR, as explained later] of 12 mg/kg mercury in maternal hair from the Faroe Islands and the no-observed-effect level (NOEL) of 15.3 mg/kg mercury in maternal hair from the Seychelles as the basis for its revised PTWI [provisional TWI]" (EFSA 2012), whose average, 14 mg/kg, was considered to be an estimate of the concentration of mercury in maternal hair reflecting exposure that would have no appreciable adverse effects in these two study populations.

"Neurodevelopmental toxicity of methylmercury in a population highly exposed from environmental sources was first recognized in the 1950s in Minamata, Japan, in association with consumption of highly contaminated fish during pregnancy. This resulted in at least 30 cases of cerebral palsy and severe developmental retardation in prenatally exposed children (Harada et al. 1968), as well as in several neurotoxic effects in highly exposed adults.

"The best available data for assessing the risk of adverse effects for MeHg are from the Faroe Islands study" (NRC 2000). Neurodevelopmental endpoints have been studied in Cohort 1 ($n = 1022$), established in 1986–1987 and Cohort 2 ($n = 182$) established in 1994–1995. "Participants in Cohort 1 performed a variety of neurobehavioural tests at age 7 and 14 years, and the investigation included clinical examinations with a focus on nervous system function" (NRC 2000).

The Japanese eat a lot of seafood, although not as much as the Faroe and Seychelle islands. The average methylmercury intake of Japanese from 1994 to 2003 is reported to be 1.18 μg/kg bw/week (8.4 μg/person/day for a body weight of 50 kg) by FSCJ [Food Safety Commission, Japan] (2005). Since the international PTWI is 1.6 μg/kg bw/week (NRC 2000). The Japanese have exceeded the international PTWI. In the Faroe Islands, the average is 36 μg/day, which is four times of that.

Fetal disorders can occur at lower concentrations. Even a mercury concentration of 15 ppm in the maternal hair has an effect on the language test. The details of the risk model will be described later.

Like precedents criteria established in other countries, the guidelines do not prohibit mercury above a certain concentration. Rather than uniform regulation, the guideline helps to consult personal consumption of seafood. Consumers may control their mercury intake by choosing seafood because of the varying levels of mercury in seafood sold on the market in Fig. 2.2. As is dangerous to use many safer drugs, we should regulate methylmercury intake by ourselves. The detailed information is disclosed on the website to help self-control.

As noted by FSCJ (2005), eating the "average" amount of seafood at average contamination levels does not pose a risk of mercury. However, some people eat seafood too much, especially in the Minamata region in ca. 1960, with a very high contamination level.

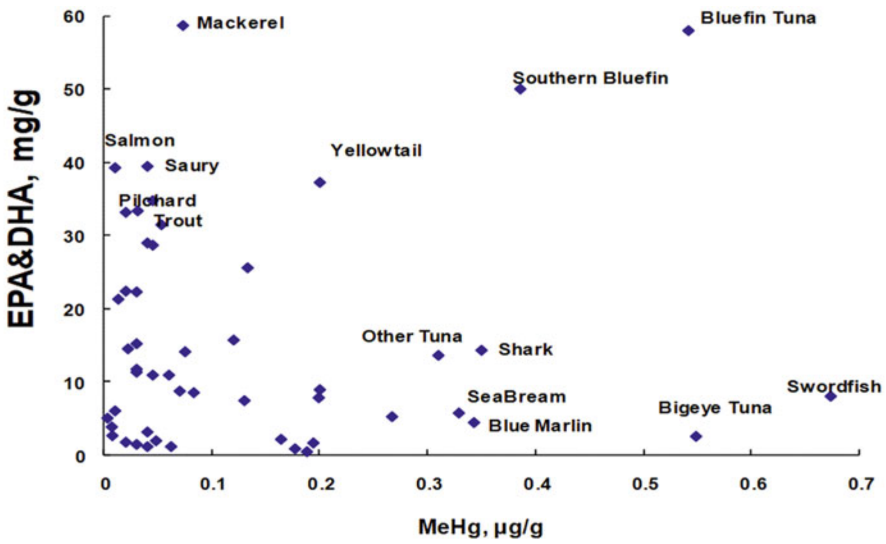

Fig. 2.2 Relationship between methylmercury concentration and EPA and DHA (unsaturated fatty acids) in fish species (Zhang et al. 2009 modified)

When considering risk, it is not enough to (1) calculate average intake and average sensitivity. (2) We need to investigate the dangerous nature of highly polluted and highly sensitive people (high-risk groups). (3) It is necessary to make a comprehensive judgment not only from a single species but from the overall diet.

Relationship between mercury levels in hair and presence or absence of various symptoms of subjects in Minamata City, and Goshoura, a part of Amakusa City, and the Ariake district as a control group has been reported (Yorifuji et al. 2009; Takaoka 2016). The data is shown in Table 2.1.

From this, the prevalence of the contaminated group is significantly higher than that of the control group, but it cannot be said that the higher the mercury concentration in the hair, the higher the prevalence rate of the contaminated group is. Regarding the number of samples for each concentration group and the frequency of people with disabilities, the P value of 5% or less in the χ^2 test is only for sensory deficits around the mouth, and all are significant if the multiple test (Bonferroni method) is considered. It can be said that there is no difference. It may be possible to discuss if the sample size is a little larger, but what can be said from Yorifuji et al. (2009) is that the number of people with symptoms is higher in the Minamata area than in the Ariake area, and the relationship with concentration is discussed. Probably the concentration was not examined at the latest time and it was not the highest concentration. If so, it is clear that people in the contaminated area are significantly more symptomatic, but we cannot say that low-level exposure is at risk for adults from these data.

Table 2.1 Relationship between the frequency of neural findings and mercury concentration in maternal hair in subjects in the Minamata/Goshoura area and the control area (Yorifuji et al. 2009). The number of cases for each symptom is calculated back from the sample size and percentage of the original article

	Samples from the polluted area					Control	P*1	P*2
Mercury concentration	~10	10~20	20~50	>50	subtotal			
Total sample number	*43*	*32*	*33*	*12*	*120*	*730*		
Bilateral sensory disturbance	10	8	13	7	38	33	19.3%	6.4E-13
Perioral sensory loss	1	5	8	4	18	1	2.7%	1.9E-15
Ataxia	12	13	15	8	48	62	26.6%	4.5E-11
Dysarthria	9	6	12	6	33	16	19.4%	3.9E-15
Tremors	6	7	7	5	25	25	32.1%	5.2E-09
Pathologic reflexes	3	1	2	2	8	14	48.9%	0.25%

[a] *P* value for dose-response relationship by χ^2-test
[b] *P* value between the polluted and control areas by Fisher's exact probability test

2.3 Risk Models of Mercury for Adults

How short-term the mercury concentration will accumulate and how long will it remain? The daily intake of methylmercury is the product of the seafood intake r (g/day) of the subject and the average methylmercury concentration h (ppm) of the seafood in the food product rh. Assuming that the bodyweight of the subject is w (kg), methylmercury accumulation x (μg) is p (μg/day), absorption rate is q (μg/day), and mercury intake other than seafood is i (μg/day). Follow the differential equation below.

$$\frac{dx}{dt} = p(fh + i) - qx$$

However, in the following, $p = 95\%$ (WHO and IPCS 1990: p 50) and $q = 1.4\%$ with a biological half-life, denoted by T_b of about 50 days (FSCJ 2005).

Assuming that the initial value $x(t)$ ($t \leq 0$) before exposure to mercury is the steady-state at $fh = 0$, $x(t) = x0 = ai/q$. The accumulated amount on day t is

$$x(t) = \frac{p}{q}\left[fh\left(1 - e^{-qt}\right) + i\right]$$

We assume that the blood concentration (μg/L) $y(t)$ when the amount accumulated in the body, denoted by $x(t)$, is expressed as $y(t) = cx(t)/w$, where $c = 0.556$.

The number of days t_c until the blood concentration becomes y_c is

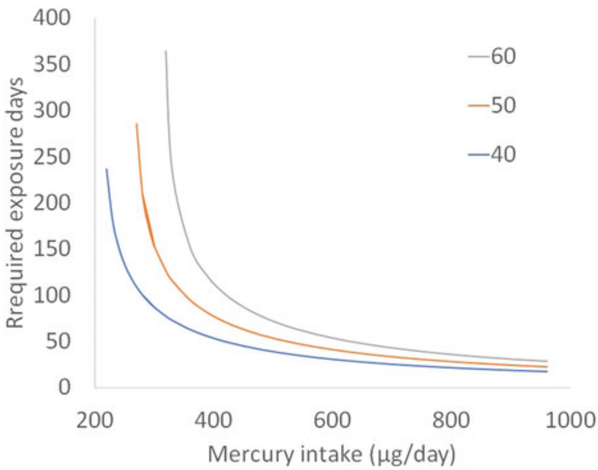

Fig. 2.3 The relationship between body weight w, mercury intake rh, and required exposure days t_c until the blood concentration reaches 200 µg/L when the bodyweight is 40, 50, and 60 kg

$$t_c = -\frac{1}{q}\log\frac{pc(rh+i)-y_c bw}{pcfr}$$

The blood concentration does not reach y_c if $y_c > pc(rh + i)/qw$.

Fig. 2.3 shows the relationship between weight w, mercury intake y_c, and required exposure days y_c when $(i, y_c) = (0, 200)$.

According to WHO and IPCS (1990), when the body burden of mercury is z mg/person, the methylmercury concentration in blood is 10z µg/L, and mercury concentration in hair is approximately 2.5z ppm. Considering the concentration (FSCJ 2005), the relationship between hair mercury concentration x ppm and the risk of onset $f(x, b, x_0, a)$ is a hockey stick model shown in Fig. 2.4:

$$f(x,b,x_0,a) = \mathrm{Min}\left[1, \mathrm{Max}\left[b, a\left(\log\frac{x}{x_0}\right)+b\right]\right]$$

where threshold concentration x_0, gradient a, and constant factor b are shown in Table 2.2.

I note that the case will happen with the probability b even though their exposure level is below the threshold x_0.

We have the data of the arithmetic mean value m and standard deviation s of the methylmercury concentration in the hair (ppm) of the households of fishery cooperative associations and others in the 1950s (Table 2.3). If the mercury concentration in the hair of subjects in each region follows a lognormal distribution, the natural log of the geometric mean and the geometric standard deviation (denoted by μ and σ, respectively) are given by

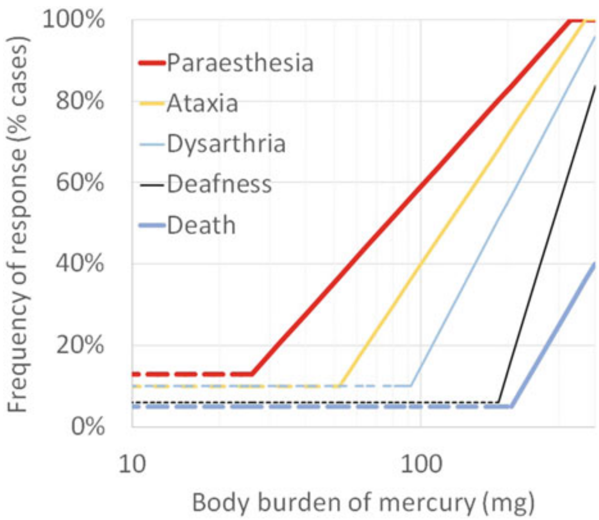

Fig. 2.4 The relationship between frequency of symptoms and the estimated body burden of methylmercury (WHO and IPCS 1990)

Table 2.2 Parameter values read from Fig. 2.4

	a	b	x_0 (ppm)
Paraesthesia	0.34	0.13	65
Ataxia	0.46	0.1	130
Dysarthria	0.58	0.1	230
Deafness	1.01	0.06	462.5
Death	0.53	0.05	512.5

Table 2.3 Methylmercury concentration in hair sampled in 1950s (Futatsuka et al. 1982). μ and σ are estimated by Eq. (2.1)

Area	Average	SD	Sample size	μ	σ
A (fishers)	66.6	44.6	199	4.01	0.61
A (total)	44.3	50.3	445	3.38	0.91
B	50.0	26.6	10	3.79	0.50
C	54.4	30.1	40	3.86	0.52
D (fishers)	43.4	28.8	75	3.59	0.60
D (total)	50.4	30.1	48	3.77	0.55
Minamata	42.2	32.8	199	3.51	0.69
E	43.8	33.2	102	3.55	0.67
F	50.8	26.5	24	3.81	0.49
G	47.2	44.2	33	3.54	0.79
H	38.8	44.3	482	3.24	0.91
J	27.4	22.9	87	3.05	0.73
K	11.4	8.7	11	2.20	0.68

$$\mu = \log\sqrt{\frac{m^4}{m^2+s^2}} \text{ and } \sigma = \sqrt{\log\left(1+\frac{s^2}{m^2}\right)} \tag{2.1}$$

I estimated the frequency of responses in each region as shown in Fig. 2.5.

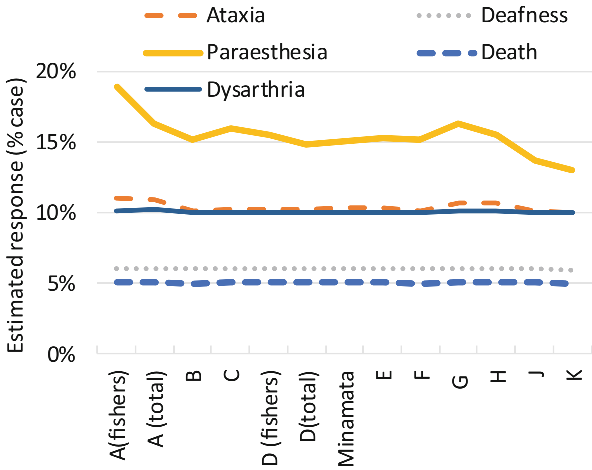

Fig. 2.5 Estimated response of Minamata disease in each region

These samples were not collected randomly but were limited to those who have voluntarily requested the test. In addition, some subjects have been delayed since the onset. The former may overestimate and the latter may underestimate the risk.

Paresthesia has been estimated to occur with high frequency in many areas. However, even without the effects of methylmercury, 13% of people develop paresthesia. Treating them all as victims includes people other than Minamata disease. It is difficult to determine the cause of the response in each case. Limiting the symptoms and identifying only those with certain symptoms of Minamata disease may exclude those who should be rescued. In the statistical context, this is a type I error that does not adopt a true hypothesis (Matsuda 2003). But if we try to save as many people as possible, we must include those who are not mercury-poisoned. This is a type II error that employs the wrong hypothesis. In the case of site J in Fig. 2.5, 13.6% of people are rescued, of which only 0.6% is attributed to mercury poisoning. The extent of the remedy depends on political decisions on how widely to apply the precautionary principle.

All affected people in the target area can be considered eligible for relief. It is unlikely that all affected people in the target area are attributable to Minamata disease, as some are affected in unpolluted areas. However, I do not know who got sick with Minamata disease. Since the prevalence rate X in the target area is higher than the prevalence rate Y in the non-polluted area, the ratio of “Y/X” of the victim is not caused by methylmercury exposure. When rescuing every symptomatic case, this will include cases not due to mercury exposure.

2.4 Risk in Children Caused by Maternal Mercury Exposure

We incorporate two kinds of uncertainties in the health risk assessment of toxic exposure, uncertainty in exposure level and uncertainty in personal sensitivity.

The best basis of a dose-response relationship for the effects of prenatal exposure on cognitive function in children is from studies in the Faroe and Seychelles islands (NRC 2000). However, the results of these two somewhat mismatched (NRC 2000). The example of Iraq used by the WHO and IPCS (1990) is not suitable for risk assessment of low-level exposure because of short-term exposure agencies and levels much higher than exposures resulting from fish consumption (Crump et al. 2000). The US Toxic Substances and Disease Registry has set the average exposure for Seychelles to NOAEL on the grounds that the hazards were not significant in the case of Seychelles, but higher exposures may not be harmful.

The cognitive dysfunction is not uniquely caused by mercury. Therefore, we estimate the exposure dose that significantly increases the prevalence of mercury-free patients. The exposure dose (BMD, benchmark dose) that causes an increase of α% (this is called "benchmark response," BMR) from the abnormal rate p_0 is used as the standard. We usually assume p_0 to be 5% and BMR to be 5 or 10%. When symptomatic subjects are salvaged, the cause of the subject's symptoms of the ratio $p_0/(p_0 + \text{BMR})$ is not due to mercury. I call $\text{BMR}/(p_0 + \text{BMR})$ the effective relief ratio.

The exposure level of so-called BMD (benchmark dose) or a lower statistical confidence bound on the BMD, abbreviated by BMDL, is considered to be the criterion for toxicity (Crump et al. 2000, EFSA 2012). As shown later (Table 2.4), the difference between BMD and BMDL is 1.5–4 folds or more depending on the type of disfunction.

The ratio of the red area to the sum of black and red areas is the effective relief ratio. If the BMR is 10%, the effective relief ratio is 2/3. However, if BMDL is used instead of BMD, the corresponding BMR is not 10%. For example, the orange area/(black and orange area) in Fig. 3 is the effective relief ratio. The higher the uncertainty of the BMD estimate, the lower the effective relief ratio. For example, as shown in $\mu(x)$ in Fig. 2.6, there is a linear relationship between exposure and performance (described later). If BMDL/BMD is 1/2, the effective relief ratio is 44%, and BMDL If BMD is 1/5, it will be 22% (Fig. 2.6).

I will explain again using mathematical expressions. A study of the Seychelles Islands by Crump et al. (2000) found that the dose of mercury in maternal hair x and child's abilities (deep tendon reflex, limb tone, Fagan visual recognition memory, Fagan attention; Mental Development Index and Psychomotor Index in 19 months after birth, Mental Development Index, Psychomotor Index, Bender Gestalt False Answer Rate, Child Behavior Checklist Total at 29 months of age, McCarthy General Cognitive Index, Preschool Language Total Score, Woodcock-Johnson's applied problems, character recognition, age at the beginning of walking in developmental stage tests, age at the beginning of speech, etc.) (logarithm as necessary).

Table 2.4 Comparison of BMD and BMDL (ppm) of mercury concentration in hair of various tests (NRC 2000 modified)

Study	End point	BMD[a]	BMDL	BMD/BMDL
Seychelles (data from Crump et al. 1998, 2000)	Bender copying errors	$>\underline{100}$	25	>400%
	Child behavior checklist	21	17	124%
	McCarthy general cognitive	$>\underline{100}$	23	>435%
	Preschool language scale	$>\underline{100}$	23	>435%
	WJ applied problems[b]	$>\underline{100}$	22	>455%
	WJ letter/word recognition	$>\underline{100}$	22	>455%
Faroe Islands (data from Budtz-Jørgensen et al. 2000)	Finger tapping	20	12	167%
	CPT reaction time	17	10	170%
	Bender copying errors	28	15	187%
	Boston naming test	**15**	**10**	**150%**
	CVLT: delayed recall	27	14	193%

[a]BMDs are calculated from the K-power model under the assumption that (p_0 = BMR = 5%)
[b]Woodcock-Johson tests

The relationship between the prevalence rate and the exposure level x is as follows. Other models have also been proposed, but the results are not so different.

As for the results in each inspection, the distribution $f(s, \mu, \sigma)$ follows the following normal distribution:

$$f(s,\mu,\sigma) = \frac{1}{\sqrt{2\pi\sigma^2}} \exp\left[-\frac{(s-\mu)^2}{2\sigma^2}\right]$$

where μ and σ are the average score and SD. Suppose this mean μ depends on the individual's mercury exposure x and other attributes C_j by the K-power model as

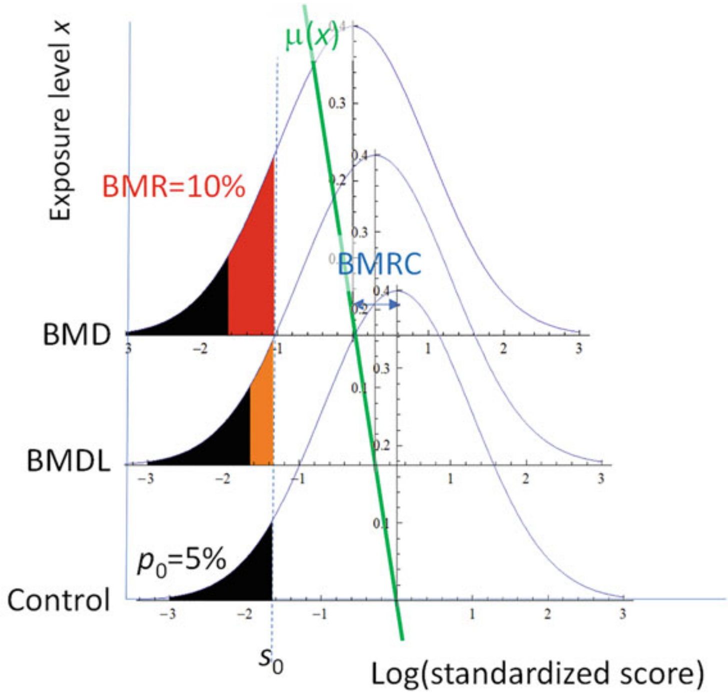

Fig. 2.6 Conceptual diagram of benchmark dose (NRC 2000 modified)

follows. The variance of scores is assumed to be constant (σ^2) regardless of x and other explanatory variables.

$$\mu(x) = (\beta x)^k + \beta_0 + \sum_j \beta_j C_j$$

Here, the explanatory variables C_j other than the mercury concentration x in maternal hair are the sex, bodyweight of the child, the income of the parent, and others.

I denote a threshold by s_0 for scores when scores fall from the bottom of the group to p_0. We usually set $p_0 = 5\%$. If the cumulative normal distribution standardized to mean 0 and variance 1 is $F(s)$, the probability that the grade will be s or less is

$$p(s) = F\left(\frac{s-\mu}{\sqrt{2\sigma^2}}\right) \equiv \int_{-\infty}^{s} f(t, \mu, \sigma)\mathrm{d}t = \frac{1}{2}\left(1 + Erf\left[\frac{s-\mu}{\sqrt{2\sigma^2}}\right]\right)$$

When I use the inverse function $(s - \mu)/\sigma = F^{-1}(p)$, I obtain $s_0 = \mu + \sigma F^{-1}(p_0)$. When I denote BMR and BMD by p_{BMR} and s_{BMD},

$$s_{\mathrm{BMD}} = \mu + \sigma F^{-1}(p_{\mathrm{BMR}} + p_0)$$

The standardized difference between BMD and s_0, BMRC $\stackrel{\mathrm{def}}{=}$ $(s_{\mathrm{BMD}} - s_0)/\sigma = F^{-1}(p_{\mathrm{BMR}} + p_0)$, is called BMRC, a given change in the mean response normalized by the SD. I note that BMRC is derived only from BMR and p_0 by the above definition. The benchmark dose corresponding to s_{BMD} is given by:

$$x_{\mathrm{BMD}} = \mu^{-1}(s_{\mathrm{BMD}})$$

where $\mu^{-1}(s) = x$ is the inverse function of $s = \mu(x)$.

BMDL is calculated from the sum of log likelihoods, denoted by L. The BMD model maximizes the likelihood ($L_{\max}$). Using the likelihood ratio test, the difference from the likelihood L(BMDL) of the BMDL model is $[L_{\max} - L(\mathrm{BMDL})]= \chi_1{}^2(0.95)$. I show BMD and BMDL for testing various fetal disorders (Table **2.4**). Crump et al. (2000) show a scatter plot of the concentration of mercury in the hair of the mother and the performance of the child, and an example of the regression model obtained from it. In this example, the maximum likelihood estimated value of the regression model $\mu(x)$ is such that BMD is 100 ppm or more (or infinity if the point estimate of β is not positive). Even if so, BMDL is usually finite because the 95% lower limit value of the interval estimation of β is positive. If the true BMD is infinite, the true rescue target ratio (BMR) will be almost *0*, and the effective relief ratio might be almost zero. If the effective relief ratio is to be calculated from the maximum likelihood estimated value, I need the information of the regression model of BMD, especially k-value, the magnitude of nonlinearity, in addition to the value of BMD and BMDL.

2.5 Type II Errors and Precautionary Principle

Safety standards are set with a sufficient margin within a practically achievable range. If the real exposure exceeds the standard, it does not immediately mean danger. On the other hand, liability in legal cases arises when a clear risk is proved, which is inconsistent with the safety standard. I would also like to note that the concept of BMD is based on the premise that symptoms will appear even if there is no influence of mercury. Therefore, I propose the "effective relief ratio" corresponding to the type II errors.

As mentioned above, there are two types of victims of Minamata disease in Japan. We note that the certified persons examined the symptoms relatively strictly and targeted those with a high probability of being affected by methylmercury, whereas the relief patients were identified based on an "estimated innocence" principle. In empirical science, the hypothesis is not supported unless a causal relationship is demonstrated or the "null hypothesis" that there is no causal relationship is statistically significantly rejected.

When considering risk, it is insufficient just to calculate (1) average intake and average susceptibility. (2) We need to examine the hazardous property of highly contaminated persons and lower susceptible persons. (3) It is necessary to judge synthetically not only from a single species but from the whole eating habits.

Although I targeted the influence on an embryo in the above analysis, if a person with an average contamination level, there is no big problem for adults. When high-concentration mercury is taken in, the problem is that health disturbance like Minamata disease might arise. Nakanishi et al. (2003) wrote that LOAEL (the lowest observed adverse effect level) of the person with the lowest mercury concentration in the blood was 0.2 ppm in the Minamata disease symptoms person. In addition, perception disorder might appear at a concentration lower than 0.2 ppm. Therefore, we used a tenfold **safety factor** and 0.02 ppm is made into allowable limit concentration. For a person with a weight of 60 kg, the allowable limit of the amount of methylmercury intake will be 30 μg/person/day, and if the Japanese average weight shall be 50 kg, an allowable limit will be 25 μg/person/day.

On the other hand, according to the National Health and Nutrition Examination Survey, the Japanese average total mercury intake is 8.42 μg/person/day (FSCJ 2005). Eighty percent of these are ingestion from seafood, and we intake 1.70 μg/person/day from other food. Most ingestion of methylmercury from other than food is negligible. Most mercury contained in seafood is probably methylmercury.

The mean total mercury intake from fish and shellfish by the Japanese was 7.4 (FSCJ 2005) or 24 μg/day (Nakagawa et al. 1997), but other studies gave different estimations. Unfortunately, the total mercury concentrations of all species are not known.

If a person eats less than 80 g of seafood per day, I assume that a person also eats seafood with an average mercury concentration. Since I did not use the average total mercury concentration for every seafood but use the concentration of similar species, the exact amount of mercury ingestion is not expressed. However, the mercury ingestion from tunas occupies most. Although there are some fish species with high mercury concentration including whales, the contribution of these species on the total intake rate of mercury is not significant because Japanese eat few average amounts of these species. In addition, the average amount of food ingested is a product of the percentage of a person who eats this species and the amount of food that is eaten by a person.

The Rio Declaration at the 1992 Earth Summit encourages precautionary measures on environmental issues and agrees on the following principles: “In order to protect the environment, the precautionary approach shall be widely applied by States according to their capabilities. Where there are threats of serious or irreversible damage, lack of full scientific certainty shall not be used as a reason for postponing cost-effective measures to prevent environmental degradation.” It does not mean that remedies should always be taken at an unproven stage. It has conditions and is said to be of concern for “serious or irreversible” effects. Usually, it is not simply “as much as possible,” but if the magnitude of the adverse effect is large, it is often said to be “as low as reasonably achievable” (ALARA).

There is a numerical standard for the tolerance of type I error regardless of the discipline. Thus, in normal empirical science, it can be said that there is a principle to avoid type 1 errors. And how large the type 1 error has an influence is not related to the rejection probability.

In contrast, the precautionary principle is considered to avoid Type II errors. However, unlike avoiding type 1 errors, there is no common quantitative standard for avoiding errors. In the first place, the probability of a type II error is not calculated unless the true probability is known. Instead, the Rio Declaration sets out the qualitative criteria for hazards as "the case of seriousness or irreversibility." However, in the example above, the probability of people responding without exposure to methylmercury was given by a hockey stick model, so we estimated type II errors.

Even if the contribution risk is low, it can be statistically significant if the number of samples is large. In that case, the effective relief ratio will decrease or even almost nothing. The extent to which it will be tolerated will be an administrative decision.

BMDL is established based on the precautionary principle as a guideline for tolerable mercury intake rate. However, in Japan, the precautionary principle is used as the reason for the relief of victims of Minamata disease for those who actually have some symptoms. Even maximum likelihood estimates are probably considered insufficient evidence to hold a company liable. Even if there is a 5% chance that mercury exposure is the cause, then liability will not be incurred. On the contrary, if there is a less than 5% chance that the null hypothesis is true, then liability may be incurred. Japan may be the only country that uses BMDL as the basis for ordering relief in court. However, in Japan, as mentioned above, the number of certified patients for which companies are responsible is limited, and the targets relieved by the government are widely applied.

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Chapter 3
Risk of Radioactive Contamination Caused by the Accident of Fukushima Daiichi Nuclear Power Plant

Tetsuo Yasutaka, Hiroyuki Matsuda, and Keisuke Kaji

Abstract The tsunami that occurred at the 2011 Tōhoku earthquake caused a severe accident with a hydrogen explosion at the Fukushima Daiichi Nuclear Power Plant. The radioactive material leaked was the second largest compared to the Chernobyl accident in 1986. The risk of carcinogenesis due to low-dose radiation exposure was concerned, and many residents were evacuated. What is important for risk science is (1) to estimate the magnitude of risk as accurately as possible; (2) to provide it as a basis for citizens' decision-making; and (3) to propose effective policies to reduce overall risk. Initially, the concentration of radioactive substances in agricultural and fisheries products sometimes exceeded the safety standards, but from around 2015, it was almost eliminated. Soil pollution does not diminish easily, but it can be dealt with by treating the soil. The food safety standard for radioactive cesium in Japan was set at 100 Bq/kg after the accident, but the actual exposure dose from food was sufficiently low. Issues will be discussed regarding how to set safety standards and the concept of risk based on the linear no-threshold model.

Keywords Linear no-threshold model · Dose-response relation · Safety standard · Carcinogenesis risk · Freedom of choice · Internal exposure

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3.1 Radiation Exposure Risk Caused by the Nuclear Power Plant Accident

On March 11, 2011, the Tohoku-Pacific Ocean Earthquake with moment magnitude 9.0 occurred in the northeastern (Tohoku) part of Japan. A massive tsunami struck the Tohoku region, with nearly 20,000 dead and missing people. A large-scale blackout including the Tokyo metropolitan area has occurred. The direct economic damage is estimated at 16–25 trillion yen (Cabinet Office, Japan 2011).

The tsunami caused fatal damage to three reactors of the Fukushima Daiichi Nuclear Power Plant (hereinafter abbreviated as FDNPP). About an hour after the earthquake, a 14 m tsunami hit the breakwater of the FDNPP. The height of the seawall was only 5.7 m, the tsunami submerged the spare diesel generators, and FDNPP lost electricity and all cooling systems.

At 14:46 on March 11, 2011, reactors 1, 2, and 3 were automatically shut down due to a large earthquake. Reactors 4, 5, and 6 were regularly maintained and not in operation during that time, but spent fuel was stored in reactor 4.

A large explosion occurred at Unit 1's external structure at 15:36 on March 12. The Unit 3 reactor building exploded at 23:01 on March 14. A loud shock and vibration were generated at Unit 4 that was not operating around 6:14 on March 15, which damaged the reactor building. Water fortunately remained in the spent nuclear fuel pool of Unit 4, which prevented heating. The meltdown occurred at reactors 1, 2, and 3 at the FDNPP.

The main cause of the release of radioactive materials was the hydrogen explosion at Unit 3 on March 14, which caused radioactive materials to diffuse into the atmosphere and deposit on the ground surface and the ocean due to rain and snow. The radioactive substances released into the environment are mainly ^{131}I, ^{134}Cs, and ^{137}Cs. The high-concentration area distributed to the northwestern part of FDNPP, and the radiation level in the area located 40–50 km northwest of the region was above 20 mSv/year (Fig. 3.1). In addition, radioactive materials have widely contaminated the area from the Eastern and Northeastern regions of Japan.

Following the spread of these pollutions, the Government of Japan set a range of 20 km from FDNPP as a restricted area on March 12. On April 22, a high-concentration range extending northwest of FDNPP was set as a planned evacuation area, and an emergency evacuation preparation area was set (Fig. 3.1). A series of evacuation orders forced more than 100,000 people to evacuate.

After that, the government started decontamination in a wide range (the range where the additional exposure dose exceeds 1 mSv), and the evacuation order was canceled sequentially from the area where the radiation dose decreased. As of January 2020, evacuation orders have been lifted except for areas called "difficult-to-return areas," but more than 20,000 people are still forced to evacuate.

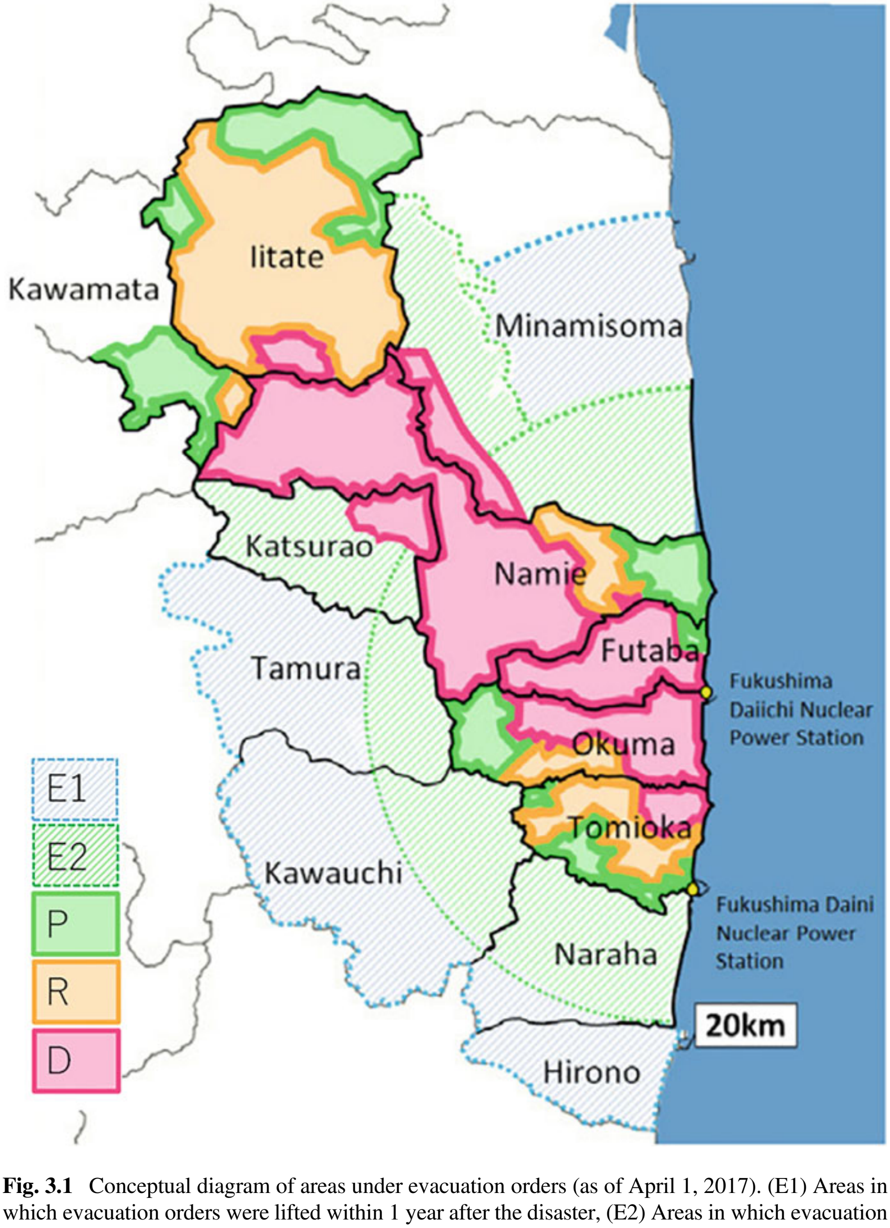

Fig. 3.1 Conceptual diagram of areas under evacuation orders (as of April 1, 2017). (E1) Areas in which evacuation orders were lifted within 1 year after the disaster, (E2) Areas in which evacuation orders were lifted between 2012 and 2016, (P) Zone in preparation for the lifting of the evacuation order, (R) Restricted residence area, and (D) Difficult-to-return zone (Reconstruction Agency, Japan 2017)

3.2 Released Nuclides and Their Behavior on Land

The physical half-life periods of ^{131}I, ^{134}Cs, and ^{137}Cs are 8 days, about 2 years, and about 30 years, respectively. Since ^{131}I has a short half-life, its concentration has dropped to about 1/1000 in 80 days after the accident, and ^{137}Cs remains in the environment as of 2020. It can be confirmed that the air dose is reduced by about 65% due to the half-life (although there is some decontamination effect) (Fig. 3.2).

Cesium deposited on the ground surface is strongly adsorbed to soil and its mobility is extremely low. From the changes over time in the concentration distribution of radioactive cesium in the soil after the accident, it was found that the depth remained in the range of 5–10 cm even after several years had passed since the accident. The radioactive cesium concentration in the soil in Fukushima Prefecture varies greatly depending on the region, but the soil concentration in the surface layer 5–15 cm immediately after the accident was about 100 Bq/kg to 100 kBq/kg (Fig. 3.3).

On the other hand, due to the strong adsorption of ^{137}Cs to the soil, transfer from soil to environmental water was limited. For example, regarding the radioactive cesium concentration in environmental water, according to the results of a large-scale survey of river water in 2017, the concentration of dissolved radioactive cesium in river water was very low at 1–10 mBq/L at many points. In addition, even considering the migration of suspended ^{137}Cs adsorbed to the sediment during large-scale flooding during heavy rainfall, the outflow rate of radioactive cesium from the basin is about 0.1–0.3% per year (Fig. 3.4).

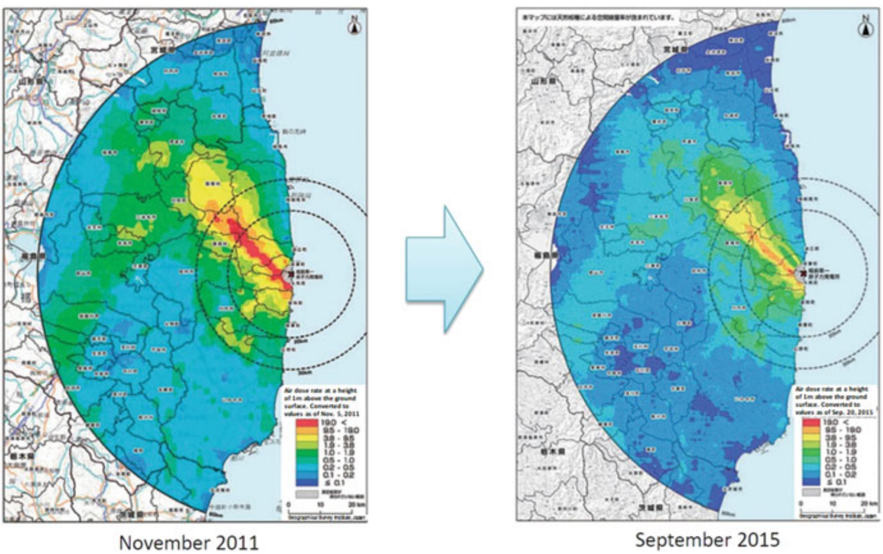

Fig. 3.2 Changes in Air Dose Rate obtained by aircraft monitoring of areas within 80 km from FDNPP. (modified from The Nuclear Regulation Authority 2011, 2015)

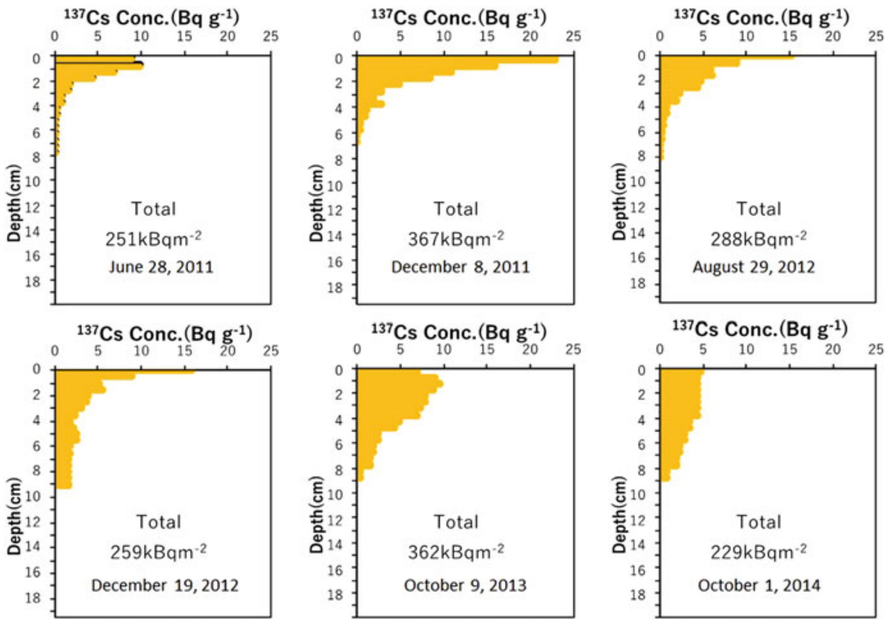

Fig. 3.3 Distribution of radioactive cesium concentration in soil in Fukushima prefecture (Takahashi et al. 2018) from June 28, 2011 to October 18

In addition, the property of strongly adsorbing to soil also affects the transfer to plants and crops. Some crops harvested in 2005 exceeded the provisional regulatory value that was set in the early stage of the accident. However, agricultural products such as rice, beans, vegetables, and fruits that had exceeded the standard for food were almost never found (Ministry of Agriculture, Forestry and Fisheries 2018). In addition, wild foods such as wild vegetables and mushrooms that exceed the standard are still found as of 2020, and the dynamics of ^{137}Cs in forests are more dynamic than in soil.

3.3 Radiation Risk Assessment and Concept of Protection

Next, consider the risks caused by radioactive substances. Human health effects of radioactive materials are mainly due to exposure to radiation (e.g., gamma rays and beta rays). As mentioned earlier, radioactive substances decay into other substances, which emit radiation. When ^{137}Cs decays, it eventually becomes ^{137}Ba, with the emission of gamma rays. To receive these radiations in the body means "exposure."

Radiation exposure includes external exposure and internal exposure. It is called external exposure to receive radiation outside the body from radioactive substances

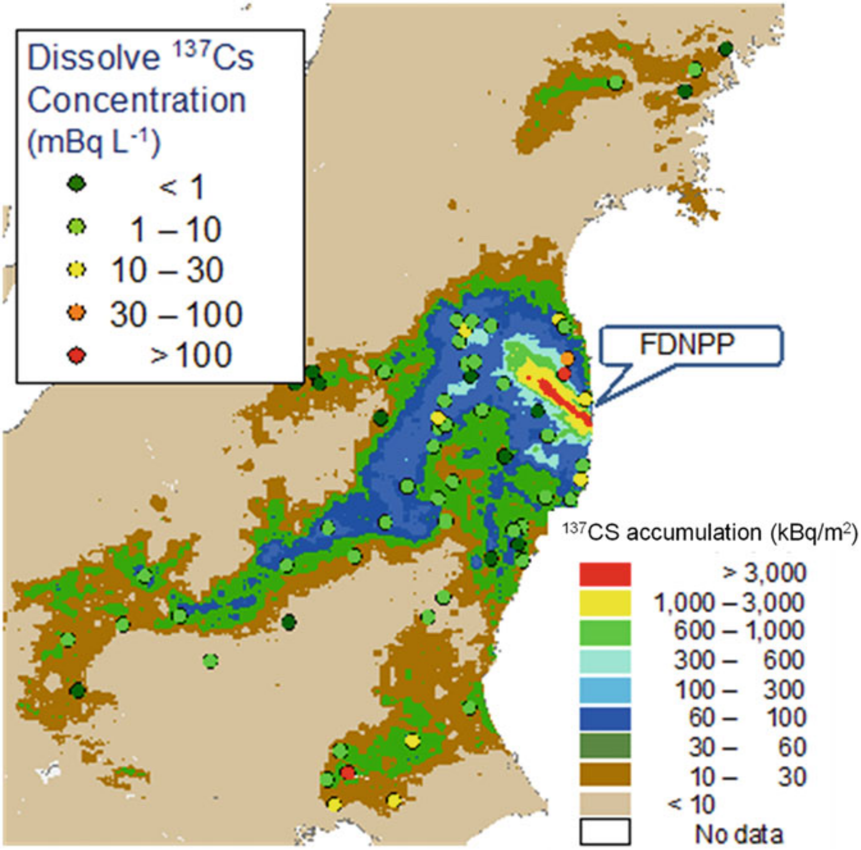

Fig. 3.4 Water sampling points (1–66) and watersheds. The watersheds were determined by the elevation map. (Revised from Tsuji et al. 2019)

existing on the ground surface or in the atmosphere. On the other hand, ingestion of radioactive substances into the body is called internal exposure.

The unit of radioactivity is the becquerel (Bq). One Bq is defined as the activity of a quantity of radioactive material in which one nucleus decays per second. In the case of ^{137}Cs, 1 Bq emits one gamma ray per second (Fig. 3.5).

On the other hand, the physical absorption of ionizing radiation dose per unit mass of one joule per kilogram of irradiated material is measured by gray (Gy). In addition, the unit of exposure dose is represented by sievert (Sv), which represents the magnitude of exposure of the living body, and is a unit of the total exposure amount due to external exposure and internal exposure. 1 Gy roughly corresponds to 1 Sv, but the effect of physically absorbed dose on the body differs depending on the type of radiation and the part of the body that received the same radiation. We usually use μSv as the exposure dose per hour and mSv as the unit of the annual exposure dose.

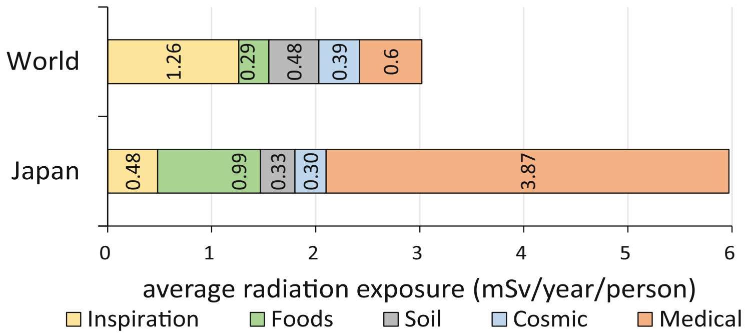

Fig. 3.5 Annual dose received by natural and artificial radiation (UNSCEAR 2008; NSRA 2011)

3.3.1 Exposure in Daily Life

Even in our normal lives, we are exposed to natural radiation from cosmic rays, food, the earth, and the atmosphere. In Japan, the average natural radiation exposure from foods, inspiration, soil, and cosmic is 2.1 mSv per year. The amount of external exposure from cosmic rays when we fly between Narita and New York is about 0.15 mSv.

In addition to natural radiation exposure, there are medical exposures such as X-rays and computed tomography (CT). Despite a large individual difference, the average annual exposure of Japanese is 3.87 mSv per year.

3.3.2 Internal Exposure

There are four types of internal exposure pathways: oral intake by food, inhalation of radioactive substances from the atmosphere (respiration), dermal intake through the skin, and penetration into the wound. Internal exposure means that the body receives radiation until radioactive substances are excreted outside the body.

The effects of internal exposure depend on the retention period in the body and the organs that accumulate depending on the type of radioactive substance. In addition, it also depends on the exposure pathways between respiration, food, and drink. Furthermore, how much radioactive material stays in the body depends on the age of the subject.

Calculating the amount of internal exposure from the amount of radioactive substances ingested into the body is called dose assessment of internal exposure. The intake is estimated for each radionuclide nuclide. The dose is calculated by multiplying intake of each radionuclide by the dose coefficient. In this dose calculation, we calculate the total amount of dose exposure (committed dose) that will be received in the future from the amount of radioactive material while it is ingested

Table 3.1 Committed effective dose coefficient (μSv/Bq) by age (ingestion) (ICRP 2012)

Age	^{131}I	^{134}Cs	^{137}Cs
3 months old	0.18	0.026	0.021
1 year old	0.18	0.016	0.012
5 years old	0.1	0.013	0.0096
10 years old	0.052	0.014	0.01
15 years old	0.034	0.019	0.013
Adult	0.022	0.019	0.013

until it is excreted outside the body. Table 3.1 shows the committed effective dose coefficient by age.

The calculation is simple. For example, the committed effective dose, i.e., exposure amount, when an adult eats 1 kg of rice with 100 Bq/kg of ^{137}Cs is 100 Bq/kg $\times$ 1 kg $\times$ 0.013μSv/Bq, where the last factor is the dose coefficient by adult for ^{137}Cs in case of oral intake. Therefore, it becomes 1.3μSv.

What about the actual internal exposure in the case of FDNPP disaster? As for internal exposure, many studies such as whole body counter, the duplicate portion method, and market basket method have been conducted, which showed that internal exposure is small. For example, at a whole body counter implemented in Fukushima Prefecture, the exposure dose of 344,565 people was measured, and 99.99% had an effective dose of less than 1 mSv per year and a maximum value of 3 mSv (Fukushima Revitalization Station of Fukushima Prefecture Government 2020).

Also, in the duplicate portion method conducted by Co-op Fukushima from 2011 to 2012, it was estimated that the internal exposure dose estimated from the diet of 100 households in Fukushima prefecture was 0.136 mSv per year even when the maximum concentration was ingested. It has been clarified that the internal exposure of citizens in the FDNPP accident was smaller than that of natural radiation.

Figure 3.6 shows the results of actual measurement of radioactive substances in each household dinner after the accident until October 2012 by duplicate portion method. At most home dining tables, radioactive cesium (Cs) is below the detection limit. Instead, natural contamination of ^{40}K was found on all samples. This is a unit of becquerel, and radioactive cesium (Cs) is about twice as high as that of ^{40}K in the unit of Sievert, but it is still sufficiently small. Contamination of radioactive cesium is much smaller than the variation of natural exposure.

3.3.3 External Exposure

External exposure is "exposure" to radiation from outside the body from radioactive substances existing on the ground surface or in the atmosphere. For the external exposure, the air dose rate (mSv/h) is measured in many sites inside and outside Fukushima Prefecture. Figure 3.7 indicate the air dose rate at the Fukushima City. The air dose rate has been increasing since March 14, 2011 after the hydrogen explosion at Unit 3 of the FDNPP. However, it has gradually decreased by the natural decay of the ^{131}I, ^{134}Cs, and ^{137}Cs.

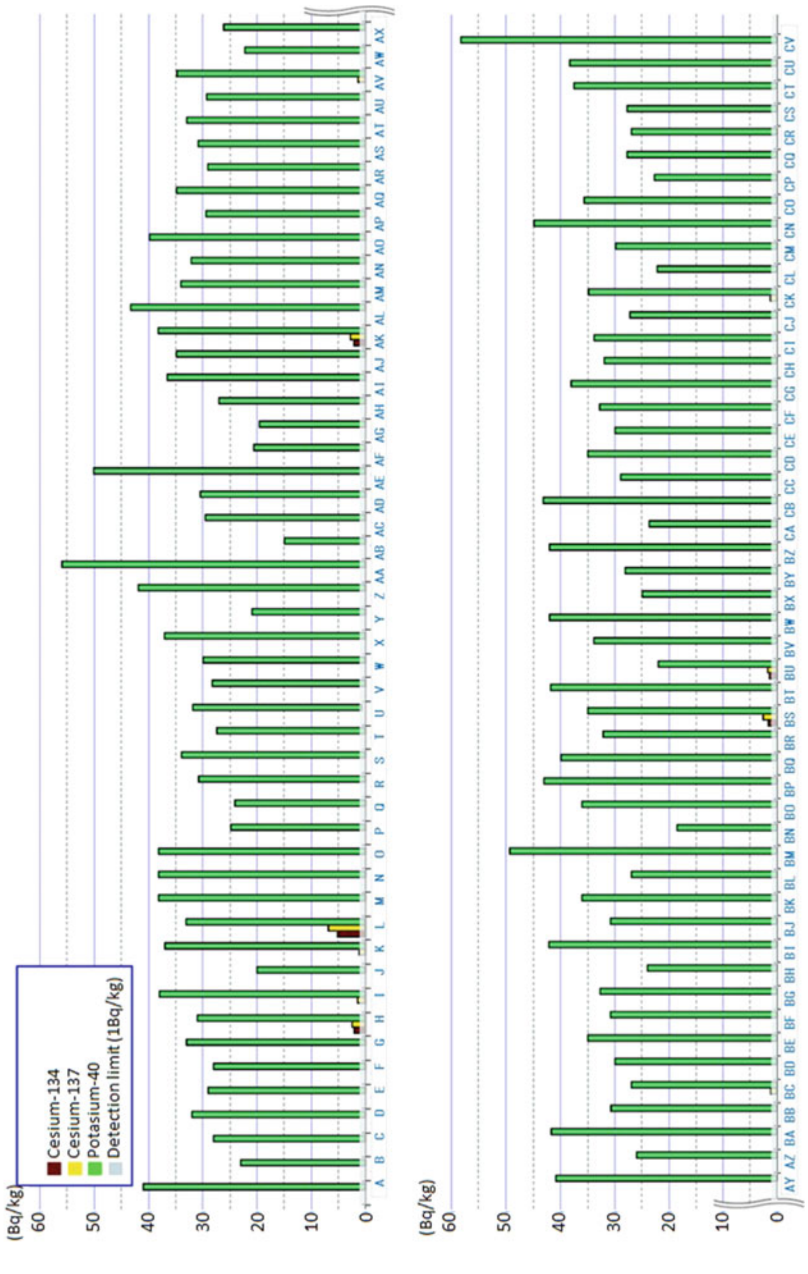

Fig. 3.6 Radioactive substance concentration in foods of households in Fukushima prefecture (modified from Co-op Fukushima 2012). The detection limit of each substance is 1 Bq/kg for the amount of ^{134}Cs, ^{137}Cs, and ^{40}K

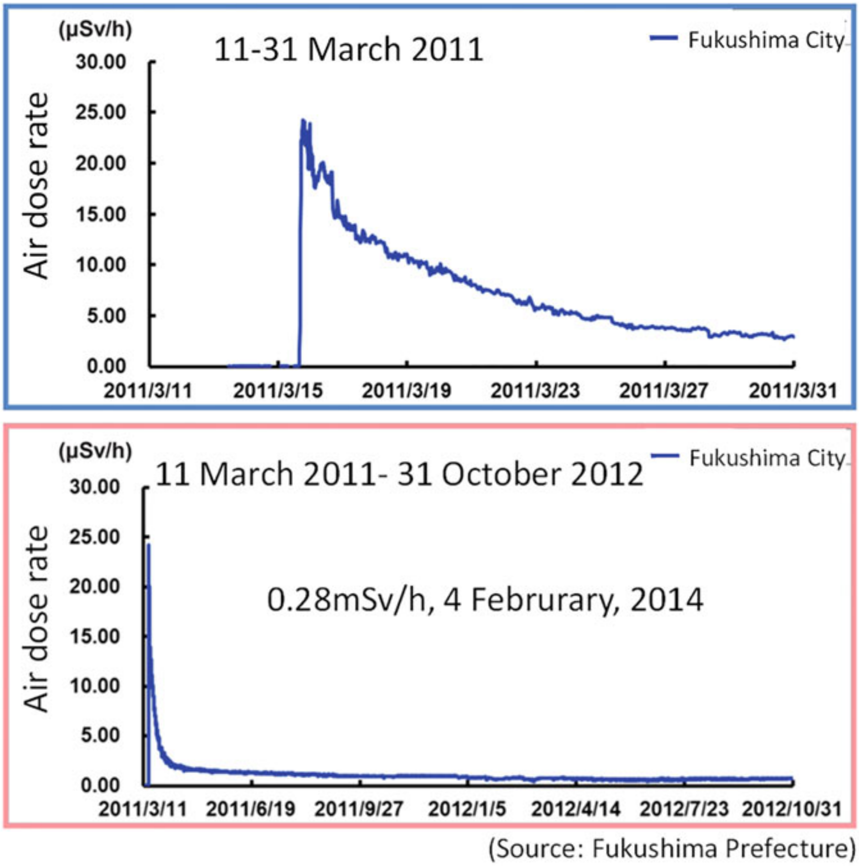

Fig. 3.7 Change of air dose rate at Fukushima City (Ministry of the Environment, Japan 2016)

The radiation dose in the atmosphere fluctuates due to the diffusion of radioactive materials and the concentration decay due to the radiation emission of the main substances ^{131}I, ^{134}Cs, and ^{137}Cs, the temporary increase during rainfall.

Daily external exposure is calculated by the following equation:

$$\sum E_e(t) = \sum [A_d(t)T_{\mathrm{out}} + A_d(t)\ S_f T_{\mathrm{in}}]$$

Where $E_e(t)$ is Daily external exposure (μSv), $A_d(t)$ is air dose rate (μSv/h), T_{out} is time spend outdoors (h), S_f is shielding factor of building (0.6), T_{in} is time spend indoors (h), t is time.

$A_d(t)$ (air dose rate) will decrease due to the half-life periods of the three substances.

$$A_d(t) = E_1 \exp[-r_1(t - t_0)] + E_2 \exp[-r_2(t - t_0)] + E_3 \exp[-r_3(t - t_0)]$$

where E_1, E_2 and E_3 are the contributions (non-negative constants) to the radiation dose from ^{131}I, ^{134}Cs and ^{137}Cs at the date of May 6, 2011; and r_1, r_2 and r_3 are –(log2)/8.02, –(log2)/754 and –(log2)/11,000 from the half-lives of ^{131}I, ^{134}Cs and ^{137}Cs, respectively. In later findings, we approximately assume that $E_2 = E_3$.

The radiation dose for each nuclide decreases exponentially, but when there are multiple nuclides, the contribution of nuclides having a long half-life becomes dominant, and the decrease becomes slower with time. Therefore, when Fig. 3.7 is expressed in a semi-logarithmic graph, $E_e(t)$ is not a straight line, but a downwardly convex curve. In any case, the radiation dose as of May 6, 2011, has already been reduced to 1.4μSv/h, which is one digit lower than the concentration immediately after the accident.

If we use the observed data until May 6, and use the regression equation after that, the cumulative radiation dose E for one year is approximately written

$$E = \sum_{t=1}^{t_1} E(t) + \frac{E_0\left(1 - e^{r_0(t_1-365)}\right)}{r_0} + \frac{E_1\left(1 - e^{r_1(t_1-365)}\right)}{r_1} + E_1(365 - t_1)$$

It was estimated to be about 1 mSv per year in Nihonmatsu City.

If hydrogen explosions, exposure of spent nuclear fuel, or opening of containment valves continue to occur after the accident, the concentration in the air is expected to increase each time. Also, if a person is not outdoors but in a wooden house 24 h a day the radiation exposure is said to be 40%, so if a person is outdoors 8 h a day, the radiation exposure would be 40% of the above.

Cumulative exposure E for 1 year is t_1 as of May 6, and is the observed value up to the present, and if the regression equation is used thereafter.

3.4 Radiation Carcinogenic Risk

Radiation exposure dose increases the risk of cancer. Radiation exposure does not lead to zero risk below a certain threshold exposure. This is called a "no-threshold" model. Basically, at an annual dose of 10 mSv, it is estimated that 114 out of 100,000 people will get cancer. Unfortunately, this number is uncertain. Originally, as shown in Fig. 3.8, the proportional coefficient was estimated from various observation examples from about 30 mSv to 1800 mSv, and the estimation error of the observed value was also quite large. In addition, there is no proof that there is a proportional relationship. There are only a few observation examples in the case of low-dose exposure of 100 mSv or less. It is definitely unverified knowledge.

The linear no-threshold (LNT) hypothesis is widely adopted for the carcinogenic risk of low-dose exposure. Although this may overestimate the risk, it is based on the precautionary principle to regulate the dose. The LNT hypothesis states that exposure to 100 mSv will increase lifetime cancer incidence by 0.17% and lifetime cancer deaths by 0.05%. However, the ICRP states that the health effects of low doses are

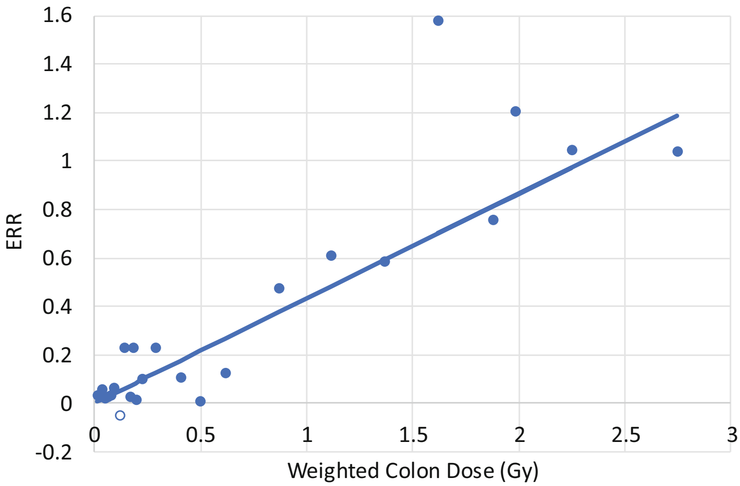

Fig. 3.8 Excess relative risk (ERR) for all solid cancer in relation to radiation exposure. The black circles represent ERR for the dose categories, together with trend estimates based on LNT model. (Redrawn from Ozasa et al. 2012)

uncertain and should not be used for virtual calculation of individual risk due to small doses over a long period of time.

3.5 Concept of Radiation Protection

The International Commission on Radiological Protection (ICRP) standard separates "planned exposure" in normal times and emergency exposure in the event of an accident. It is commonly said that there is a "safety myth" that assumes that nuclear power plants will not cause a critical accident. The nuclear power plant policy clearly describes what to do when a severe accident occurs rather than other issues such as thermal power plant. ICRP had a distinction between planned exposure and emergency exposure before the FDNPP accident. In addition, radiation exposure is inevitable in daily life without accidents. This is called planned exposure and can be divided into (1) occupational exposure and (2) medical exposure. On the other hand, when a serious accident such as re-criticality may occur, an emergency treatment is needed. The FDNPP Accident is truly a serious emergency, and the radiation exposure of the residents corresponds to the emergency exposure.

The upper limit in normal times (planned exposure situation) recommended by ICRP (2007) is set to 1 mSv per year excluding medical exposure and natural radiation exposure (Fig. 3.5). The emergency standard is a reference level that is applied to emergency exposure situations and existing controllable exposure situations, and it is said that the value depends on the circumstances surrounding the exposure situation. In other words, they have not set a numerical threshold.

However, at doses higher than 100 mSv, there is an increase in fatal effects and a statistically significant risk of cancer, so a baseline value above 100 mSv should not be set annually. The standard value of public exposure in an emergency is set between 20 and 100 mSv per year. This reference level of public exposure applies to situations where the measures to reduce exposure are infeasible. Occupational exposure is said to be unlimited for life-saving activities.

What we expected from the discussion on the radioactive contamination standard value of food was what kind of thinking should be used to determine the safety standard that can be implemented in case of emergency. When pollution continues, there are many things that are not achievable if the standard value is set based on the idea of normal times. Therefore, even if we take a step back, in an emergency, we will have to think about standards that can be reliably implemented and achieved in the situation. On the other hand, establishing the same "loose" criteria from normal times as in emergency is contrary to the "as low as reasonably achievable (ALARA)" policy. ALARA was first used by the ICRP in 1997 but is now widely used in risk management in other fields.

Normal times standards are not the maximum numbers to avoid risk. The less pollution is the better, so we try to keep it as low as reasonably achievable. Therefore, if it is below the standard value, there is sufficiently no risk, and if it exceeds the standard value, it does not mean "immediately dangerous." It is often said that, for example, the standard value of arsenic in foods differs between Europe and Japan. It is expected that there will be many Japanese rice that exceed Western standards. Because the rice grown in the United States is not so contaminated, probably because of the difference in the arsenic concentration, strict standards do not hinder agriculture in the United States. In another example, the bluefin tuna mercury content does not even meet the standards of other seafood in Japan. However, it is recognized as a luxury item. The reason is that it is not eaten every day. This concept is valid from the principle that the health risk depends on the total dose, not the number of times food is eaten above the standard value.

3.6 Environmental Remediation Process

The environmental remediation activities for the radioactive materials released from the FDNPP accident started from 2012. This activity included "decontamination process," "temporary storage site," "interim storage facility," and "final disposal site outside the prefecture," as shown in Fig. 3.9.

Decontamination is the process of removing radioactive cesium from agricultural land, forests, road, houses, etc. (see Fig. 3.10). Areas with an additional annual exposure dose of 1 mSv/year (equivalent to 0.23μSv/h) were designated as areas to be decontaminated. However, due to the huge area of the forest in Fukushima (over 70%), only 20 m of the forest from the living area was decontaminated.

After decontamination, soil and waste were packed into the containers and moved to temporary storage sites (these are to be kept for 3–9 years). The number of

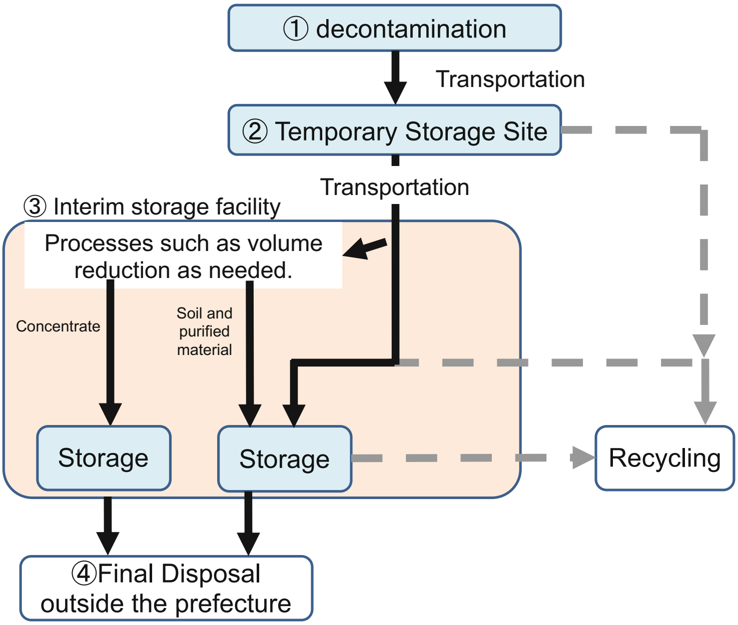

Fig. 3.9 Conceptual diagram of environmental remediation activities for the radioactive materials released from the FDNPP accident. (Modified from Yasutaka 2020)

temporary storage sites in Fukushima Prefecture are over 1000 between 2015 and 2017. In order to store these large volume of contaminated soils for 30 years, the interim storage facility occupied about 16 km^2 and was built in Futaba Town and Okuma Town where are an area adjacent to the FDNPP. Furthermore, the agreed policy was that within 30 years of starting the interim storage, final disposal of the contaminated soil and waste will take place outside the Fukushima Prefecture until 2045. However, almost nothing has been decided on for final disposal sites outside the prefecture, and site selection and consensus building with stakeholders will be required in the future.

Finally, the estimated cost of decontamination was 5.8 trillion yen. This cost did not include the cost of final disposal. The year 2045 is 25 years away, but we need to set a firm path for our generation to solve this challenge, rather than leaving it to the next generation.

3.7 Example of Risk Management: Food Safety

Since the accident at the Chernobyl Nuclear Power Plant in 1986, Japan has regulated the import of foods containing radioactive substances by setting provisional limits on imported products, but when the FDNPP accident occurred, there

Fig. 3.10 Decontamination process in Fukushima: (**a**) grassland (IT-01-P0041), (**b**) roof and buildings (IT-01-P0034), (**c**) road (FB-01-P0012), (**d**) temporary storage site (IT-02-P0056). (Ministry of the Environment, Japan 2016)

were no legally based restrictions on the domestic production of foods. Therefore, the Food Safety Committee of the Ministry of Health, Labor and Welfare of Japan (MHLW) proceeded with the work to determine the standard value while issuing a notification that the provisional regulatory value is 500 Bq/kg after the accident. As a result, the committee reported a very strict standard value of 100 Bq/kg for marine products, and after April 1, 2012, the regulation of radioactive substances in food was legally enforced under the Food Sanitation Law.

The standard value of 100 Bq/kg is much stricter than that of other countries, even as a standard value in normal times. The provisional regulation value of 500 Bq/kg immediately after the earthquake in Japan was already stricter than the standards of Europe and the United States and the International Food Standards Commission (CODEX) as shown in Table 3.2. The standard value of 100 Bq/kg further lowered it.

As mentioned above, it depends on the country whether strict standards do not interfere with industrial activities. From a radiation health risk perspective alone, the standards are as good as low, and every country sets their standard as low as reasonably achievable. However, the standard value of Japan is lower than that of other countries except Belarus. The health goal of 1 mSv per year has already been

Table 3.2 Food safety standard value of radioactive cesium of countries

	Cs-134 + Cs-137 (Bq/kg-wet)
Codex	1000 *
EU	1250
USA	1200
Hong Kong, Philippines, Vietnam Malaysia,	1000
China	800
Thai, Singapore,	500
South Korea, Chinese Taipei,	370
Japan	500 → 100
Belarus	80

[a]Codex standard includes S-35, Co-60, Sr-89, Ru-103, Ce-144, and Ir-192

met, but the committee determined a very strict standard due to unrealistic assumptions about how often to eat contaminated food.

Similar problems occurred in Europe after the Chernobyl nuclear power plant accident. Many foods in a wide range of Europe exceeded the standard value for radioactive cesium of foods in these countries. In 1986–1987, the contamination level of 150 kBq/kg in reindeer meat, 40 kBq/kg in lamb, 30 kBq/kg in freshwater fish, 1350 Bq/kg in goat milk, 12 kBq/kg in mushroom, and 650 Bq/kg in milk were detected as the maximum value of each item in Norway. Until then, the standard value for general food was 600 Bq/kg, but for reindeer, mushrooms, and freshwater fish, the standard value was raised to 6000 Bq/kg (Liland et al. 2009). Since the frequency of eating game meat will not be so high, it was expected that the internal exposure level is sufficiently safe, i.e., <1 mSv per year, even if the standard value is relaxed.

On the contrary, the standard value of Belarus is stricter than that of Japan. This is considered to be the difference in the frequency of eating contaminated food, rather than the difference in the allowable internal exposure. It is rare that radioactive cesium is actually detected on the Fukushima dining table.

One of the reasons is that Japan has a low food self-sufficiency rate of 40%. Furthermore, the percentage of people who eat foods produced in Fukushima prefecture will be lower. The food mileage indicator encourages the extent of local production for local consumption. Ironically, the low self-sufficiency rate is one of the reasons that the exposure risk of Fukushima citizens is significantly different from that of Belarus.

In the case of the food standard value of radioactive cesium (Cs), it is assumed that half of all foods are contaminated to the limit of the standard value and the other half is completely uncontaminated. In that case, the standard value is set so that the annual exposure dose is 1 mSv or less. It is a very unrealistic assumption to determine the standard value that half of all foods are contaminated.

It is often said that if someone moves from the vicinity of the evacuation zone to Osaka, where there are many granites on earth because you are worried about external exposure, the amount of external exposure to natural radiation will increase. Radioactive cesium exceeding the standard value was detected in wild mushrooms from 10 prefectures including Aomori, Nagano, Yamanashi, and Shizuoka in 2012, in which ^{137}Cs was detected but ^{134}Cs was not detected. It was speculated that this is not due to the FDNPP accident, but due to the nuclear bomb tests by the USSR and the USA in the 1960s. In other words, we have become concerned about what we used to sell and what we used to eat.

3.8 Give Consumers "Freedom of Choice"

Based on the findings at the end of May 2011, we stated that the risk of radiation exposure is not so high. However, there are many consumers who want to avoid any risk. What scientists need to do is to provide scientific resources to the public, rather than making social choices for themselves. Scientists should say a factual proposition, not a value proposition. What science may say a value proposition is called "naturalistic fallacy" (Wilson et al. 2003).

It is not a good idea for consumers to take away the "freedom of choice" to avoid low concentrations of food. Conversely, "freedom of choice" to support the production areas should be guaranteed too. It is important to provide information and give consumers choices. A food courier company sells agricultural products and seafood that are selected with more stringent criteria than the standard value. They also have a contract with a farmer in the disaster area and sell foods produced in the disaster area that is called "foods for supporting disaster area." A single courier company provides both products.

Consumers who buy food in supermarkets rarely know who produces it. If they bought foods produced by farmers near FDNPP area but did not acknowledge these farmers, the consumers may choose foods produced in other areas to avoid radiation risk. However, a courier company that contracts with the farmers in Fukushima hesitates to abandon the farmers.

We also note that the former product is called "comfortable without concern" instead of "safe." Since it is the government's view that everything below the standard value is safe, those with stricter standards do not mean safer. In Japan, objective scientific safety and subjective "comfortable without concern" are distinguished. "Anshin" in Japanese and "anxin" in Chinese mean believing that the outcomes are threat-free or "to express confidence in an outcome" (Gupta and Sharman 2009).

Many people are worried that food near the NPP is radioactively contaminated, but the level of contamination is low. Therefore, accepting these low levels of radioactive contamination will allow consumers to purchase and eat these foods and support these farmers and fishers. By doing so, consumers can economically support farmers and fishers who are suffering from the Fukushima nuclear power plant accident and radiation exposure.

On the contrary, it has been pointed out that evacuation increased the mortality rate (Murakami 2019). It is important to think about the risk trade-off that when one risk is avoided, another risk may increase.

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Chapter 4
How to Assess Ecological Risks of Trace Metals in Environment

Yuichi Iwasaki and Hiroyuki Matsuda

Abstract Ecological risk assessments for chemicals are fundamental to understand and manage their adverse impacts on the environment. In this chapter, we introduce methods to assess the ecological risks of chemicals with a particular focus on trace metals such as zinc and copper. In freshwater ecosystems which are most threatened than other ecosystems at a global scale, trace metal contamination is a long-standing concern worldwide. A typical source of the trace metal contamination is active/inactive (legacy) mines, and many of those mines have been causing different levels of environmental impacts. Here, we introduce three types of methods to assessing ecological risks and impacts of trace metals, and their advantages and limitations. First, comparing the measured concentrations of metals with environmental quality benchmarks such as water quality standards is a useful screening-level approach. Second, by performing toxicity tests with field-collected water samples, the whole toxicity to aquatic organisms (e.g., algae, crustaceans, or fishes) is directly examined. Third, field surveys of biological groups can directly capture the ecological consequences of metal exposure to aquatic populations and communities. Typical biological groups used for biological assessment in streams and rivers are periphyton, benthic macroinvertebrates, and fishes. The simplest study design for the field survey is comparing aquatic populations and communities at two river sites upstream and downstream the inflow of mine discharge. If ecological risks of a chemical are highly concerned, the countermeasures to reduce the environmental concentrations are ideally required. In Japan, the environmental water quality standard for zinc in freshwater was established to be 30 μg/L to protect aquatic populations and was derived based on a chronic toxicity value for a macroinvertebrate (mayfly) species. However, the results of field surveys in multiple rivers suggest that dissolved zinc concentrations of more than twice the standard (70–115 μg/L) has little effect on macroinvertebrate richness and thereby that the Japanese water quality standard is somewhat overprotective for the protection of the macroinvertebrate richness.

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Accumulated findings clearly indicate that field survey can be used to reasonably infer “safe” concentrations in the natural environment.

Keywords Metal contamination · Metal bioavailability · Hazard quotient · Uncertainty factor · Water quality benchmark

4.1 Ecological Risk Assessment for Chemicals in General

The number of chemicals registered in the Chemical Abstracts Service (CAS) is over 140 million and a recent study estimated that over 350 thousand chemicals have been globally registered for production and use (Wang et al. 2020). Chemicals are essential and beneficial for our life, but the historical experience clearly indicates that their impacts on ecosystems as well as human health (see Chap. 1) can be critical. To avoid and minimize such adverse impacts, we need risk assessments to examine, e.g., if new chemicals entering the market are safe (e.g., the Toxic Substances Control Act (TSCA) in the United States and Registration, Evaluation, Authorization and Restriction of Chemicals (REACH) in the European Union), if there is a risk of concern for any chemical in the natural environment, and if the discharge from a given point source (e.g., manufacturing plant) is affecting aquatic organisms inhabiting in the downstream river.

Tiered approaches are commonly used for ecological risk assessment (ERA) for chemicals. As a screening-level (first) assessment, ecological risks are examined by a simple and generally conservative (or safe-side) way with a minimum (available) toxicity dataset. Such screening-level ERA is typically performed by dividing predicted environmental concentration (PEC) by predicted no-effect concentration (PNEC; Fig. 4.1):

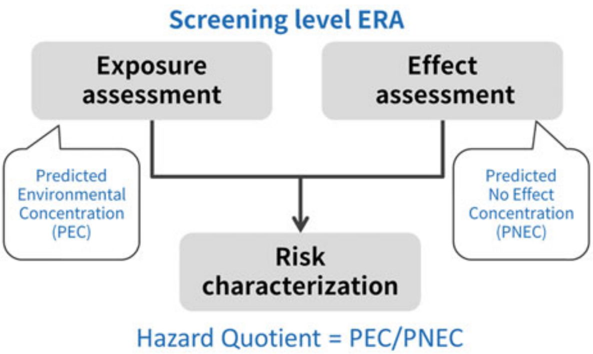

Fig. 4.1 Screening-level ecological risk assessment (ERA)

Table 4.1 Test durations and endpoints in acute and chronic ecotoxicity tests

Biological groups	Type	Time	Endpoints
Algae	Acute	$\leq$72 h	Growth Inhibition
	Chronic	$\geq$72 h	Growth Inhibition
Crustacean (cladocerans)	Acute	48 h	Immobilization (mortality)
	Chronic	21 days	Fecundity, growth, immobilization (mortality)
Fish	Acute	96 h	Survival
	Chronic	>7 days	Hatchability, fecundity, growth, survival

Durations and endpoints shown here are some examples based on, e.g., OECD guidelines for testing chemicals, and they can vary depending on test species, life stages of species tested, and regulatory jurisdictions

$$\text{Hazard quotient} = \frac{\text{PEC}}{\text{PNEC}}$$

If the hazard quotient (HQ; Suter 2007) is well below than 1, the risk is *not* at the level of concern and further testing or risk reduction measure is not required. If the HQ is close to or higher than 1, the risk may be at the level of concern and further detailed assessment is required. Although those are typical examples, how we interpret the HQs may change depending on, e.g., regulatory jurisdictions, methods used, and empirical evidence available.

Here, let us pretend that we are performing the nationwide ERA for a chemical in freshwaters. In the exposure assessment (Fig. 4.1), the maximum concentration of a chemical observed in the nationwide water quality monitoring may be used as the "safe-side" PEC. However, given that the maximum value is not a statistically robust estimate (if the sample size increases, the maximum value can increase), use of quantiles such as the 0.95 quantile (95th percentile) is a better alternative. If the water quality monitoring data is lacking, exposure models that simulate watershed hydrology and water quality are often used to derive the PEC.

In the effect assessment (Fig. 4.1), the PNEC is derived from the results of ecotoxicity tests. In toxicity tests, by exposing a given biological species (typically, an algal, crustacean, or fish species) to a range of concentrations of a chemical of concern, individual-level effects (endpoints) such as those on survival, growth, and fecundity (e.g., number of spawned eggs) are investigated. Although a tremendous number of biological species are present in freshwaters, a limited number of species such as *Raphidocelis subcapitata* (a microalgal species), *Daphnia magna* (a planktonic crustacean), and *Pimephales promelas* (fathead minnow; a cyprinid fish species) are commonly used in ecotoxicity data. Acute (short-term) or/and chronic (long-term) toxicity are examined in toxicity tests, in which the testing periods and endpoints vary depending on the species tested (Table 4.1). Median lethal concentration (LC50) and 50% effective concentration (EC50) are usually estimated in acute toxicity tests, while no observed effect concentration (NOEC; the maximum concentration at which no statistically significant effect is observed), and EC10 (or EC20) are derived as "safe" concentrations in chronic toxicity tests.

Table 4.2 Hypothetical examples of the derivation of predicted no-effect concentration (PNECs) for a chemical by applying different uncertainty factors (Forbes and Calow 2002)

Case	Toxicity values (mg/L)			Uncertainty factor	PNEC (mg/L)
	Algae	Crustacean	Fish		
Case 1		100 (acute)		1000	0.1
Case 2		10 (chronic)		100	0.1
Case 3	15 (chronic)	10 (chronic)	50 (chronic)	10	1

PNEC is the concentration below which any unacceptable impacts on ecosystems will most likely not occur and is rather loosely defined. To derive a PNEC, there are two major sources of uncertainty, lack of understanding, and natural variability, the latter of which cannot be avoided because it is the inherent property of ecosystems (Forbes and Calow 2002). Generally, we need to derive a PNEC based on very limited toxicity data (say, only a single acute toxicity data). In other words, we need to extrapolate the results of limited toxicity tests performed in the laboratory to ecological impacts in actual environments. Do you think it is really possible? To address this uncertainty, in ERAs, the minimum toxicity value is commonly divided by the uncertainty (so-called, application, assessment, or safety) factors (UFs). Although scientific evidence supporting the use of such factors has not been fully gained, there is a long history of the use in ERAs. For instance,

1. If you have only a single acute toxicity value (e.g., LC50), a UF of 1000 is applied to take into account the acute to chronic ratio, variation in interspecies sensitivity, and lab-to-field extrapolation.
2. If you have only a single chronic toxicity value (e.g., NOEC), a UF of 100 is applied to take into account variation in interspecies sensitivity and lab-to-field extrapolation.
3. If you have at least three chronic toxicity values from each of the three biological groups (see Table 4.1), a UF of 10 for lab-to-field extrapolation is applied to the minimum toxicity value.

By assuming that a crustacean species is most sensitive among the three biological groups, Table 4.2 shows some hypothetical examples for the derivation of PNECs. Although those are simple examples, the derived PNECs depend on the data availability. Compared to Case 3 in Table 4.2, PNECs for Cases 1 and 2 are underestimated (i.e., indicating overprotection). Thus, the interpretation of PEC exceedance of PNEC should be assessed based on the availability of information (i.e., how reliable a PNEC or PEC is; Chapman 2018).

When the screening-level ERA suggests that the risk may be at the level of concern, more ecotoxicity data or environmental monitoring data are needed to be acquired to more accurately estimate the PNEC or PEC. For instance, once ecotoxicity data are obtained for many biological species (say, 5–10 species or more), species sensitivity distribution (SSD) may be used to estimate a PNEC (Posthuma et al. 2002). The SSD expresses a set of toxicity values such as EC10 and NOEC, as a statistical distribution (e.g., log-normal distribution) and has been used to estimate the hazardous concentration for 5% of the species (HC5) that can be used to derive a PNEC by applying a smaller UF (usually, 1–5). SSDs are now commonly used to derive water quality benchmarks such as environmental water quality standards/criteria in many regulatory jurisdictions.

Importantly, the exceedance of a chronic toxicity value such as NOEC or EC10 for a chemical does not necessarily mean that the corresponding species will disappear at a site or area of concern. Even if the concentration for a chemical exceeds the chronic toxicity value for a species, it does not mean that no individual of the species reproduces or survives at all. As explained in the theory of sustainable fisheries, many organisms become resilient in their attempts to maintain population persistence when the population size decreases. Even if the survival and reproductive rates decrease, the species are not so fragile that their populations continue to decrease before the extinction. However, if there is the effect of a chemical beyond the resilience of the population, the population will not recover and will continue to decline. The chronic toxicity values such as EC10 and NOEC, at which almost no or small effects on survival or/and reproduction are expected, should be somewhat protective to guarantee the population persistence.

To address such issue of unclear ecological consequences based on the chronic toxicity values, Kamo and Naito (2008) developed a population-level SSD approach. In the approach, first, threshold concentrations leading population extinction for individual species are generally estimated by incorporating concentration-effect relationships into life-history parameters. In the case of zinc, the concentration that guarantees the persistence of 95% of the species population (called the population-level hazardous concentration of 5% of species) was estimated to be 107 μg/L, while HC5 based on a conventional (individual-level) SSD based on NOECs was 27 μg/L (Fig. 4.2).

4.2 Metal Pollution and Ecological Impacts

Environmental pollution with trace metals such as copper, zinc, cadmium, and lead is a worldwide concern in terms of the protection of human health and ecosystems (Luoma and Rainbow 2008). Before getting down to the point, the two terms are to be discussed. First, although the discrimination between "contamination" and "pollution" is not often done even in scientific literature, contamination is the presence of a pollutant and pollution is "contamination that causes adverse biological effects in the natural environment" (Chapman 2007). Also, the term "heavy metal" is very commonly used in the field of environmental science but is surprisingly ill-defined (surprisingly many definitions are available; Duffus 2002). Because of this, the use of heavy metal has been occasionally but severely criticized (Duffus 2002; Hodson 2004); We like the title of Hodson's paper, "Heavy metals - geochemical bogey men?"). There are several alternatives such as a classification of metallic elements based on the periodic table (s-, p-, d-, or f-block; see Duffus 2002 for more details). Those alternatives are unfamiliar, but this should not be a reason not to use them and

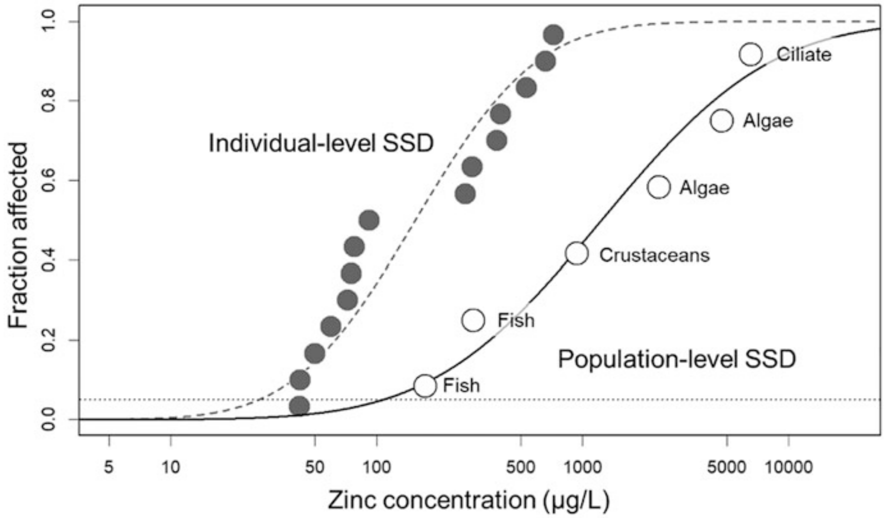

Fig. 4.2 Individual-level (broken line) and population-level (solid line) species sensibility distributions (SSDs). Filled and open circles indicate NOECs for 15 biological species and the threshold concentrations leading to population extinction for 6 biological species, respectively. A horizontal dotted line indicates the fraction affected of 5%

it is important to recognize that "heavy metal" is not scientifically valid term. In this chapter, the term "metal" or "trace metal (metal generally found in low concentration)" is used to express those such as copper, zinc, cadmium, and lead.

Numerous studies have demonstrated aquatic (freshwater, marine) and terrestrial ecosystems affected by trace metal pollution (Adriano 2001; Luoma and Rainbow 2008). Among the different ecosystems, freshwater ecosystems such as streams, rivers, and lakes harbor diverse species and provide important ecosystem services, but these ecosystems are currently being stressed by various anthropogenic impacts (Flitcroft et al. 2019; Reid et al. 2019). Freshwater ecosystems are the most threatened at a global scale based on the Living Planet Index—that is, the average rate of change over time across a set of species populations (WWF 2018). The freshwater Living Planet Index, representing over 3000 populations of 880 species, showed an 83% decline. Water pollution is one of the significant causes, and others include habitat modification including instream flow modification, fragmentation, climate change, invasive species, and so on.

In freshwater ecosystems, a typical source of metal pollution is mining. There are many inactive (legacy) mines worldwide, as well as active mines, and many of those mines have been causing different levels of environmental impacts. Historic disasters caused by active/inactive mines sometimes occurred and recent examples include the Gold King Mine spill, Colorado, USA (US Environmental Protection Agency 2018) and the collapse of the Fundão tailings dam, Minas Gerais, Brazil (do Carmo et al. 2017), both of which occurred in 2015. Particularly, the collapse of Fundão dam was "the biggest environmental disaster of the world mining industry,

Fig. 4.3 Photo of an acidic river downstream a closed mine in Japan. Even though the riverbed was covered by the orange-colored iron deposition, a few species of macroinvertebrates such as nemourid stonefly larvae (Plecoptera) were present

both in terms of the volume of tailings dumped and the magnitude of the damage" (do Carmo et al. 2017). In contrast, interestingly, the clear ecological effects of the Gold King Mine spill, that is, those on fish populations and macroinvertebrate communities in rivers, were not found despite the devastating fact that the Animas river temporally turned yellow likely due to the chemical reaction of iron and aluminum with the river (US Environmental Protection Agency 2018). In addition to such extreme events, acidic mine drainages (AMDs) are perennially released from individual mines and often contain high levels of trace metals such as cadmium (Cd), copper (Cu), lead (Pb), nickel (Ni), and zinc (Zn) (Fig. 4.3). Regardless of whether AMDs are treated or not, ecological impacts of trace metal contamination in freshwaters caused by mine drainages are a long-standing concern worldwide and have been extensively studied (Luoma and Rainbow 2008; Namba et al. 2020). In this regard, ecological risk assessments have a crucial role to provide fundamental information about the predicted or observed impacts and thereby how the metal contamination should be managed.

4.3 Factors Affecting Toxicity of Metals

Elevated metal concentrations in environmental media such as surface waters in rivers can cause direct and indirect ecological impacts. Although the comprehensive and detailed mechanical understanding of metal toxicity is still lacking, it has been well demonstrated that waterborne metal toxicity depends on the metal bioavailability that is affected by several water quality parameters including major ions (e.g., Ca and Mg), pH, and dissolved organic matter (Adams et al. 2020). For instance, major ions such as Ca and Mg compete with trace metals at biotic ligands and thereby increased those concentrations mitigate the toxicity of trace metals. US EPA chronic water quality criteria for zinc depend on water hardness (2.497 $\times$ Ca (mg/L) + 4.118 $\times$ Mg (mg/L)) and those at water hardness of 20 and 100 mg/L, are 30 and 120 μg/L, respectively (US Environmental Protection Agency 2002). Also, the binding of trace metals to dissolved organic matter can reduce the metal toxicity (i.e., reduced toxicity with increased dissolved organic matter).

Historically, the influence of major ions, pH, and dissolved organic matter on the metal toxicity was a critical issue when assessing ecological risks of trace metals because results of toxicity tests were too variable to derive a reliable PNEC and also because the total, dissolved (filtered), or free-ion concentration of metals measured is not necessarily a good predictor of toxicity. To address this issue, the biotic ligand models (as known as BLMs), which assume that toxicity occurs by the accumulation of metals on biotic ligands (e.g., gills of fish), have been developed and tested extensively for many trace metals. An excellent example is that the acute toxicity measures (LC50s) based on the estimated concentrations of Ni and Cu bound to gills of a fish species (fathead minnow) were constant when Ca concentrations increased, whereas those based on the total and free-ion concentration were not (Meyer et al. 1999). Note that, in BLMs, the influences of those water quality parameters on the metal bioavailability vary among trace metals (OECD 2017) and often among biological species of concern.

Although empirical evidence has been accumulated for use of BLMs, there are several issues to conclude that BLMs should be used to predict trace metal exposure and environmental impacts on diverse aquatic species in general. First, in environments, aquatic organisms can accumulate metals from their food and often sediment in addition to water, and the relative importance of the individual exposure routes to the overall bioaccumulation and resulting toxicity is often uncertain. Cation may be required particularly when the concentrations of trace metals in water, sediment, and food are not correlated. Second, the accumulated evidence is largely based on laboratory testing with a very limited number of species. Based on the state-of-the-art knowledge, ease of sampling and analyzing, and the circumstance of study sites of concern (e.g., little variation in water quality parameters), use of simple exposure predictors such as total and dissolved concentrations of metals can be adequate.

4.4 Methods to Assessing Ecological Risks of Metals Caused by Mine Discharge

In this section, we discuss three types of methods to assessing ecological risks and impacts of trace metals caused by mine discharge as well as their advantages and limitations. To this end, we focus on the trace metal pollution in freshwater, particularly, streams and rivers. Given that many active/inactive mines are located in mountainous areas, the streams and rivers are the typical environmental media receiving the mine discharges. Also, such location characteristic can lead to negligible impacts of other anthropogenic factors such as agricultural/urban pollution and land-use changes, and thereby causal inference based on field surveys should be more straightforward. For the simplicity and reasons explained in the previous section, we will use dissolved concentrations of trace metals as a predictor for effects.

4.4.1 Ecological Risk Assessment Based on Measured Concentrations of Metals

Let us pretend that we are interested in assessing the ecological impacts of mine discharge in a river. A likely first step is to assess whether or not ecological risks are of concern in the river by performing a screening-level ERA. By collecting and analyzing water samples (preferably multiple times) at several study sites in the river, dissolved concentrations of trace metals of concern (e.g., Cd, Cu, Pb, Zn) and other relevant water quality parameters such as water hardness are measured. Water samples for dissolved metals analysis are filtered (0.45-μm mesh size) at the field sampling and analyzed in the laboratory. Then, the hazard quotient (HQ; see Sect. 4.1) can be calculated by the measured concentration of a metal divided by a PNEC or relevant water quality benchmark. The measured concentration may be the averaged or maximum concentration of multiple samples collected, e.g., at different times of the year. For example, if the long-term effects of mine discharge, which perennially flows into the river and has relatively constant concentrations of metals, are of concern, comparing the averaged concentration of metals with chronic water quality benchmarks such as USEPA chronic water quality criteria (U.S. Environmental Protection Agency 2002) would be appropriate. If episodic effects are of concern (e.g., historic disasters mentioned above), comparing the maximum concentration with acute water quality benchmarks may be more relevant.

The ecological risks can be assessed by examining the magnitudes of HQ values calculated for individual metals (see Sect. 4.1 for the typical interpretation). Importantly, even though all the HQs calculated for individual metals are below 1, the effects of metal mixtures may be of concern. Examining if the sum of HQs exceeds 1 is a potentially useful way to assess the ecological risks of metal mixtures but the threshold of 1 may be too protective (Iwasaki et al. 2020). In either way of assessment, the limitations should be paid attention. Because water quality benchmarks are typically conservative and not absolute, the final decision-making should not be made based only on those benchmarks (Chapman 2018). If ecological risks are concerned, performing further detailed assessments are recommended (see Sects. 4.4.2 and 4.4.3).

4.4.2 *Toxicity Testing with Field-collected Waters*

By performing toxicity tests with field-collected water samples, the whole toxicity to aquatic organisms (e.g., algae, crustaceans, or fishes) is directly examined. This type of toxicity testing is more commonly applied to facilities' effluents and is called whole effluent toxicity (WET) testing (Grothe et al. 1996). Test durations and endpoints in Table 4.1 may be used. An advantage of this approach is to directly examine the whole toxicity of a water sample, while the HQ approach needs to predict the overall toxicity by summing up HQ values of individual metals or chemical substances. If an unmeasured metal or chemical substance significantly contributes to the overall toxicity, the HQ approach cannot take it into account but the toxicity testing approach can do. Another advantage is to be capable of identifying the water sample characteristics causing toxicity and the procedure is called toxicity identification evaluation (TIE; Grothe et al. 1996). However, it is uncertain how the observed toxicity links to effects in the field.

4.4.3 *Field Surveys of Biological Groups*

In contrast to the other two methods described above, an appealing advantage of field surveys of biological groups is to directly capture the ecological consequences of metal exposure to aquatic populations and communities. Typical biological groups used for biological assessment in streams and rivers are periphyton, benthic macroinvertebrates, and fishes, which have different advantages (Barbour et al. 1999). Periphyton primarily consists of algae, supporting riverine food webs as primary producers. Their life cycles are relatively short, are easy to sample, and are often highly sensitive to short-term physical and chemical disturbances. Benthic macroinvertebrates are relatively sedentary, have variable life cycles, and comprise diverse sets of species with a wide range of sensitivities to trace metals (Iwasaki et al. 2018; Rosenberg et al. 2008), being useful for assessing relatively local-scale and long-term cumulative effects. Fishes are relatively long-lived and mobile, being good indicators of longer term and broad-scale effects. In addition, because fishes are important resources for our food and recreational and commercial fishing, they are more valued than other biological groups by many local human communities.

Which biological groups (periphyton, benthic macroinvertebrates, and fish) are better to be surveyed in assessing metal impacts on aquatic populations and communities in rivers? A recent systematic review of a total of about 200 published studies worldwide concluded that (1) benthic macroinvertebrates have been most frequently used (>60% of studies), that (2) correlations between responses of the different biological groups were often low, and that (3) abundance (number of individuals) and richness (number of taxa or species) metrics of macroinvertebrates were generally more responsive to changes in metal contamination level than those of periphyton or fishes (Namba et al. 2020). These results suggest that, although it is important to survey multiple biological groups for comprehensively understanding the responses of aquatic populations and communities to metal contamination in rivers, benthic macroinvertebrates (mainly aquatic insects) could be a reasonable first choice to detect the ecological impacts of metal contamination. In this systematic review, studies only investigating the accumulation of metal in aquatic organisms were not included. In addition to the ecological impacts, if you are interested in metal accumulation in, e.g., fishes, for human health, the investigation on such aspect is further required.

By establishing the appropriate reference sites with similar physicochemical characteristics other than concentrations of trace metals and comparing the populations and communities at the contaminated and reference sites, we can infer the field impacts caused by the mine discharge. The simplest approach is comparing aquatic populations and communities at two river sites upstream and downstream the inflow of mine discharge (called, the upstream-downstream comparison). If the inflow of mine discharge is relatively low compared to the river discharge and thereby the physicochemical characteristics are similar between the two sites, the upstream-downstream comparison is probably the best to directly examine the ecological impacts of the mine discharge. For instance, if effects of concern are not observed by the upstream-downstream comparison, it is probably not necessary to examine the impact at further downstream river sites.

Further, to investigate the ecological impacts of mine discharge at multiple downstream sites or in a certain section of the river, a useful approach is to establish multiple study sites in the river and reference sites with similar physicochemical characteristics in a nearby uncontaminated river (e.g., a similar-sized river within the same basin) with low or background-level concentrations of metals. The reference sites may be established in different river basins as long as physicochemical characteristics other than concentrations of trace metals are similar. Then, aquatic communities are compared between multiple contaminated and reference sites. As an example of such a study design, the results of the macroinvertebrate survey performed in a northern Japanese river (Iwasaki et al. 2020) are shown in Fig. 4.4. Even though the sum of HQs based on US EPA criteria for Cd, Pb, Cu, and Zn exceeded 1 at the contaminated sites (1.7–7.4; particularly the exceedance of dissolved concentrations of Cd and Pb were apparent), the number of taxa and individuals of metal-sensitive mayflies (Fig. 4.4), as well as other macroinvertebrate metrics, were similar between contaminated and reference sites. These results have demonstrated that the assessment based on the sum of HQs can be overprotective in terms of the prediction of the impacts on aquatic communities in the field, and also

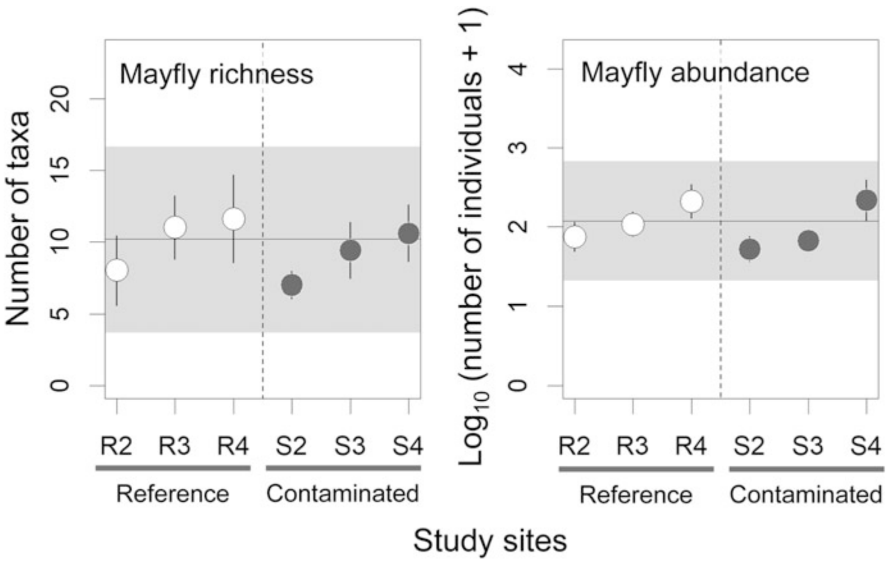

Fig. 4.4 Mayfly (Ephemeroptera) richness and abundance at reference (R2 to R4) and contaminated (S2 to S4) sites. Error bars indicate 90% confidence intervals calculated from 5 samples collected at individual sites. Gray areas are 90% prediction intervals calculated from means for the three reference sites. Data are from Iwasaki et al. (2020)

highlighted the importance of performing field surveys to better understand ecological impacts.

4.5 Summary with Some Management Perspectives

If ecological risks of a chemical are highly concerned, the countermeasures to reduce the environmental concentrations are ideally required. In Japan, environmental water quality standards are used as PNECs to evaluate the individual-level ecological risks, and the nationwide effluent standards are typically established or strengthened to reduce the environmental concentrations if the major sources of the chemicals are industries. However, because the water quality standards are established largely based on the laboratory toxicity data, interpretation of the exceedance needs some caution. For example, in Japan, the environmental water quality standard for zinc in freshwater was established to be 30 μg/L in 2003 to protect aquatic populations and derived based on the minimum toxicity value, that is, chronic toxicity to a macroinvertebrate species. However, results of the macroinvertebrate surveys in multiple Japanese rivers suggest that zinc concentrations of more than twice the standard (70–115 μg/L) has little effect on six macroinvertebrate taxon richness (mayfly richness is shown as an example in Fig. 4.5) and thereby the water quality

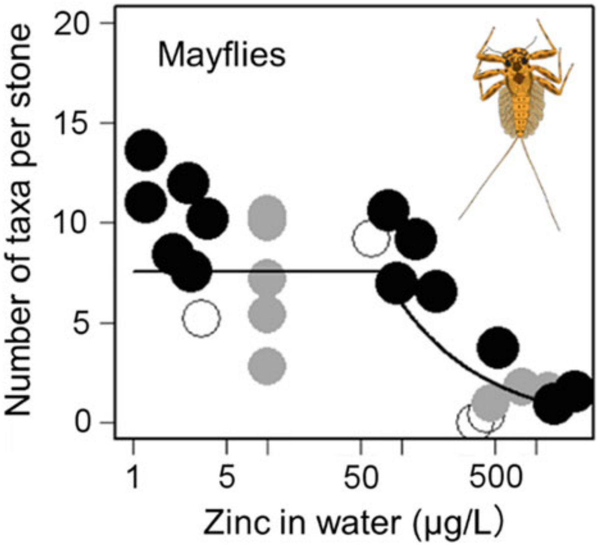

Fig. 4.5 Relationship between dissolved zinc concentrations in river water and mayfly richness (number of taxa) in Japan (Data are from Iwasaki et al. 2011). The solid line is the estimated threshold response. Different colored dots indicated different sampling rivers. Mayflies (Ephemeroptera) are often reported to be sensitive to metal contamination

standard is somewhat overprotective for protection of the macroinvertebrate richness. Furthermore, these results indicate that field survey can be used to reasonably infer "safe" concentrations in the natural environment. Interestingly, the "safe" concentrations estimated from the macroinvertebrate data are close to the population-level HC5 of 107 μg/L (Fig. 4.2).

The nationwide effluent standard was lowered from 5 to 2 mg/L in 2006 to maintain and reduce the environmental concentrations of zinc in Japan. The primary zinc sources for river sites with elevated zinc concentrations were estimated to be legacy mines (17%), and industrial point source and/or municipal wastewater treatment plants (58%; Naito et al. 2010). The latter sources indicate that many zinc-elevated sites are located in urban areas where other physical, chemical, and biological factors such as flow alternation, organic pollution, and invasive species likely affect aquatic communities. In such environment, controlling a single factor (e.g., zinc concentration) may be ineffective. Based on field survey data on macroinvertebrates, Iwasaki et al. (2018) provide empirical evidence that, because macroinvertebrate communities are severely affected and macroinvertebrate species susceptible to metal pollution should be sparse or absent in organic-contaminated rivers (5-day biochemical oxygen demand (BOD) of >3 mg/L), the reduction in zinc concentration by the effluent regulation may not be a first choice in such rivers for the recovery of lotic macroinvertebrates. A similar issue can occur at zinc-elevated sites affected by mine discharge because the mine discharges generally have elevated concentrations of multiple metals. Therefore, it would be essential to perform more integrated management rather than the regulation of individual chemicals to effectively restore aquatic ecosystems as well as employing multiple assessment approaches including water quality measurements, toxicity testing, and field surveys to better understand the ecological impacts and their causes.

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Chapter 5
Impact of Reactive Nitrogen and Nitrogen Footprint

Azusa Oita, Ichiro Nagano, and Hiroyuki Matsuda

Abstract Nitrogen is essential for all lives since reactive nitrogen (all nitrogen except for nitrogen gas) constitutes protein and nucleic acid. Synthesized ammonia, chiefly used for nitrogen fertilizers, greatly supports human food production for the growing population. The growing use of synthetic fertilizers and fossil fuels has increased reactive nitrogen emissions to the environment, leading to adverse effects on human and ecosystem health. This global issue of maximizing the benefits of reactive nitrogen while minimizing nitrogen pollution is one of the key issues for the twenty-first century. To communicate this nitrogen issue to stakeholders in different industries including farmers and consumers, the nitrogen footprint (NF) has been developed as an indicator to quantify direct and indirect reactive nitrogen emissions throughout the lifecycle of goods and services of our consumption. We introduce methodologies of the NF models in relation to other environmental footprints and demonstrate three applications of nitrogen footprint models. The feed-sensitive NF model has been developed as a bottom-up approach and applied to fish and seafood analysis with two sets of parameters called virtual nitrogen factors (VNFs) for the world and Japan. Using the Japanese VNFs, effects of dietary changes to the food NF of Japan and possible reduction scenarios for our food choice were assessed. The global NF model has been constructed as a top-down approach and applied to assess 188 countries in 2010 using multi-region input–output analysis to trace international supply chains. These NF models contribute to the development of sustainable food systems and integrated nutrient management addressing trade-offs between different nitrogen pollutants and other environmental issues.

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5.1 Agricultural Revolution and Human Alteration of the Nitrogen Cycle

5.1.1 What Is the Impact of Reactive Nitrogen?

Nitrogen is a limiting element of plant growth and all organisms need nitrogen as an essential nutrient to form protein and nucleic acid to live. Abundant atmospheric nitrogen (N_2, composing 78% of the air we breathe) has a triple bond and is inert. Therefore, in natural conditions, only a very few of the species on the earth, such as Rhizobium bacteria in the root nodules of legumes, and lightning can convert it into reactive forms like ammonium (NH_4^+) and nitrogen oxides (NO_x). This transformation of nitrogen is called **nitrogen fixation** and it produces **reactive nitrogen** (all nitrogen chemical species except for N_2), a source of plant-animal-fungi interaction. In the natural **nitrogen cycle**, some part of NH_4^+ is converted to nitrates (NO_3^-) by nitrifying bacteria, then denitrifying bacteria convert NO_3^- back to nitrogen gas. However, reactive nitrogen can cause harmful impacts on ecological systems when the systems receive much more reactive nitrogen than their requirements (Galloway et al. 2003). Besides natural nitrogen fixation, human activities accelerate nitrogen fixation by legume cultivation, fossil fuel and biomass combustion (unintentionally producing NO_x as a consequence), and industrial ammonia (NH_3) synthesis.

The excess reactive nitrogen in the environment contributes to many environmental and health problems (United Nations Environment Programme (UNEP), 2014; Sutton et al. 2019b). For example, nitrogen enrichment causes increased occurrences of harmful algal blooms, coastal dead zones, and biodiversity loss of freshwater and coastal water systems. NH_3 pollution leads to eutrophication, soil acidification, and direct toxicity in organisms, decreasing species richness and diversity. NH_3 and NO_x emissions to the air are linked to respiratory and cardiovascular diseases. Nitrous oxide (N_2O) is a greenhouse gas, which has 300 times more potent than carbon dioxides (CO_2). These reactive nitrogen effects link each other because reactive nitrogen molecules can be converted to any other nitrogen forms and move through the environment. The same reactive nitrogen atom can contribute to multiple ecological and human health effects in the air, in terrestrial ecosystems, in freshwater and marine systems, and on human health. This sequence of effects is called the **nitrogen cascade** (Galloway et al. 2003).

5.1.2 Agricultural Revolutions and Increased Reactive Nitrogen

Humans began domestication of crops and animals between ca. 11,000 and 5000 years ago (Smith, 2005). Means of nitrogen input to agricultural fields were limited to legume cultivation and use of manure until nitrogen discovery and the beginning of identifying specific nitrogen compounds in the eighteenth century, followed by the discovery of the basis of the nitrogen cycle in the nineteenth century (Galloway et al. 2013). At the end of the nineteenth century and the beginning of the twentieth century, natural sources of nitrogen fertilizers, including guano and mineral NO_3^- deposits, were used in addition to manure and legume cultivation (Galloway and Cowling 2002). In those days, fossil-fuel combustion became the primary energy source instead of biofuels (e.g., wood), which enabled sufficient energy supply for industrial manufacturing, electricity, and transportation in return for NO_x emissions.

The **Haber-Bosch process**, an energy-intensive process to convert hydrogen (H_2) and nitrogen (N_2) to NH_3 under high pressures and high temperatures, was invented in the early twentieth century. In the 1950s, the Haber-Bosch process started to expand rapidly to produce synthetic fertilizers for food production to sustain the growing population (Galloway et al. 2013). In the period of 1961–2010, thanks to industrial nitrogen fertilizer production and fossil-fuel energy, global production of vegetal proteins has increased threefold due to new crop strains highly dependent on reactive nitrogen input, together with inputs of other fertilizers, water, pesticides, and other technologies of the green revolution (Lassaletta et al. 2014; Sinclair and Rufty 2012; Tilman et al. 2002). Currently, in some regions, particularly in China, reactive nitrogen input to field exceeds the appropriate amount to improve yield, whereas some regions such as Sub-Saharan Africa need to address nitrogen deficiencies (Mueller et al. 2012). The crop production growth provided relatively cheap grain and surplus to regional food requirements to allow intensification of livestock farming, resulting in a large increase of per-capita meat and dairy production (Sutton et al. 2013). In addition to agricultural use, 20% of Haber-Bosch NH_3 in 2005, comparable to the amount to NO_x emissions from fossil-fuel combustion, is used as a raw material to create enormous numbers of industrial products other than synthetic fertilizers (e.g., nylon, plastics, rubber, resins, drug, explosive, and pesticide) (Galloway et al. 2008; Gu et al. 2013).

Where does all the reactive nitrogen input to agricultural and industrial production go? Considering the full supply chains from new nitrogen input to final food products, agricultural **nitrogen use efficiency (NUE)** is on average less than 20%, while industrial NUE is more than 90%. The rest of the agricultural reactive nitrogen input and much of industrial products after use are eventually lost to the environment (Gu et al. 2013; Sutton et al. 2011, 2013). Efforts for estimating the damage cost of reactive nitrogen are undergoing (Compton et al. 2017; Sutton et al. 2013). An attempt by Compton et al. (2011) estimated damage to ecosystem services associated with productivity, biodiversity, recreation, and clean water as \$2.2–56 per kg N,

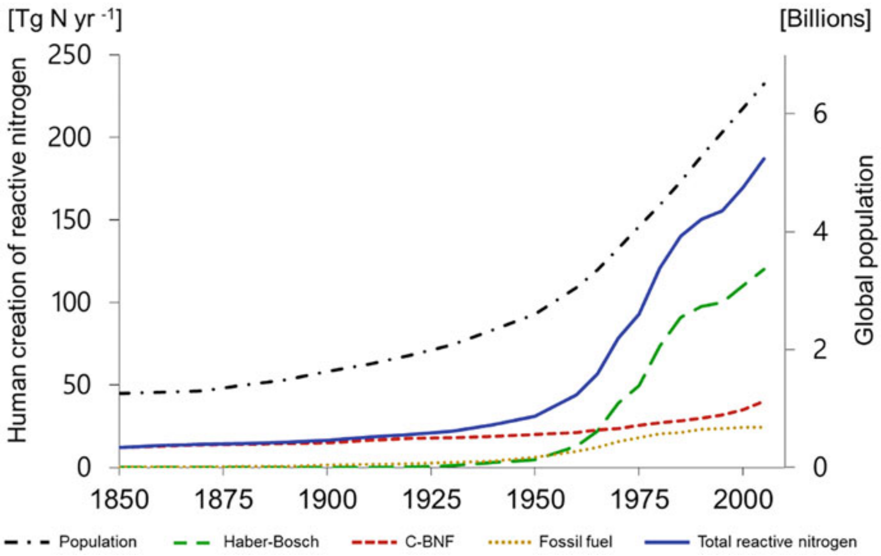

Fig. 5.1 Trends of reactive nitrogen creation and the global population based on Galloway et al. (2003). C-BNF: Cultivation-induced biological nitrogen fixation

while costs of ozone and particulate damages in relation to respiratory health were estimated as $28 per kg NO_x-N.

On a global scale, human-induced nitrogen fixation in the period of 1850–1950 was constant at ca. 12 kg N cap^{-1} year^{-1}, and the total amount increased from ca. 12 Tg (10^{12}g) N cap^{-1} year^{-1} to ca. 31 Tg N cap^{-1} year^{-1} with the growth of population (Fig. 5.1, Galloway et al. 2014). From 1950 to 1980, as agriculture become industrialized and intensified, per-capita reactive nitrogen creation had rapidly increased to approximately 30 kg N cap^{-1} year^{-1} (ca. 121 Tg N year^{-1}) in 1980. From 1980 to the current, per-capita reactive nitrogen creation remained at approximately 30 kg N cap^{-1} year^{-1}. In 2010, of the 413 Tg N annual global nitrogen fixation to all terrestrial and marine ecosystems, human activities are responsible for 210 Tg N (Fowler et al. 2013). For terrestrial ecosystems, in which the majority of human-induced nitrogen fixation takes place, reactive nitrogen input has tripled from approximately 58 Tg N year^{-1} by natural terrestrial ecosystems and approximately 5 Tg N year^{-1} by lightning.

The global nitrogen cycle is one of the nine **planetary boundaries** for critical processes to maintain the Earth system functioning, but it has been at high risk as the current human-induced nitrogen fixation is far more than the planetary nitrogen fixation boundary of 62 Tg N year^{-1}, which was set in view of nitrogen runoff to surface waters (De Vries et al. 2013; Rockström et al. 2009; Steffen et al. 2015). Producing more food while minimizing nitrogen pollution is the **global nitrogen challenge** (Houlton et al. 2019). How should we address this global nitrogen challenge? In agro-food systems and industrial systems, integrated efforts to increase

NUE should be taken by the production side (Davidson et al. 2015). At the same time, consumers have choices of what to buy and to eat. In order to show this consumers' role in the nitrogen issues, indicator tools have been developed as explained in the following subsection.

5.2 Nitrogen Footprint Development and Footprint Indicators

5.2.1 Footprint Concept and Its Applications

The **nitrogen footprint (NF)** is a consumption-based indicator to show the link between our consumption activities and nitrogen pollution. The NF quantifies the total amount of reactive nitrogen potentially lost to the environment as a result of an individual, an organization, or a country's resource consumption (Leach et al. 2012). The NF includes "**virtual nitrogen,**" all reactive nitrogen emissions before our intake of food or our use of commodities, as well as "real nitrogen" emissions, e.g., from consumer-level waste after use and sewage. It is one of the extensions of the footprint concept.

The footprint concept was first developed by Canadian researchers in the 1990s (Rees 1992; Wackernagel and Rees 1996) as the **ecological footprint** (EF) by imagining if a city was covered by a film that prevents inflows and outflows of substances. Do we use more things than we produce in the city area's ecosystems? Do we generate sewage and waste more than the environmental carrying capacity of the city area to absorb? The EF quantifies the available amount of productive land and sea area (biocapacity) and the amount of area we require to meet our demand for crop production, grazing, wood production, fisheries, CO_2 uptake, and building (EF). The current operational standards for the ecological footprint (Global Footprint Network 2009) and the most recent description of the accounting methodology and results at the time of writing (Lin et al. 2018) are available on the websites. If the EF is more than the biocapacity, it means that the potential impacts of our demands, or footprint, overshoot ecological sustainability. Since our EF is larger than the earth's biocapacity, humanity's resource consumption from January 1, 2020, to August 22, 2020, exceeds earth's capacity to regenerate those resources in 2020 (www.footprintnetwork.org/).

Following the EF, the water footprint (WF, Hoekstra and Hung 2002), carbon footprint (CF, e.g., Høgevold 2011), NF, biodiversity footprint (Lenzen et al. 2012), and many other environmental, economic, and social footprints have been developed to quantify human load on the earth and different dimensions of sustainability. The EF methods to convert demand or emissions to areas in a common unit have been debated (e.g., Kitzes et al. 2009) and many later developed footprints use mass-based or volume-based units (Čuček et al. 2012). Since the carrying capacity is not always clear for each footprint, most of the footprints only measure the demand-induced footprint part.

Footprint tools can be used in **life cycle assessment (LCA)** of a product or service, the economy of a state, a region, or a world, and a level in-between product and economy-wide (see Guinée and Lindeijer 2002 for details of LCA). In the general LCA methodological framework, there are four phases of the investigation: clear goal and scope definition, inventory analysis for all inputs and outputs associated with the life cycle of the product of study (or other study targets), impact assessment of the identified resource use and emissions generated from the product, and interpretation of the results (Fang et al. 2014). System boundaries are often set as cradle-to-grave (from raw material extraction through product use and disposal) and cradle-to-gate (from raw material extraction to factory gate). Product level LCA covering environmental impacts is standardized in the International Organization for Standardization (ISO) 14,040 series (www.iso.org/standard/37456.html). Further assessment for sustainability can extend its scope from environmental indicators to economic and/or social indicators (Guinée et al. 2011).

While LCA generally uses a set of indicators to assess multiple impacts, one footprint indicator often focuses on one environmental impact (e.g., carbon footprint for global warming). Thus, a combination of different footprints including NF as a "footprint family" (Galli et al. 2012, further integrated by Fang and Heijungs 2015) to measure broader impacts can be used to complement LCA with less data and simpler calculation. On the other hand, from an LCA perspective, the NF, which sums all reactive nitrogen into a single account, needs to be converted to different accounts to show eutrophication potential, acidification potential, global warming potential, and so on. There is an ongoing debate for LCA-compatible footprints and the footprint family (Einarsson and Cederberg 2019).

Two types of approaches have been applied to calculate environmental footprints in the literature: bottom-up and top-down approaches. The advantages of the bottom-up approach (e.g., WF for specific pasta sauce and peanut chocolate candies by Ridoutt and Pfister 2010) are simplicity of calculation and sensitivity for accessing product level difference. It is a widely used approach in practical business. However, this approach does not capture the entire supply chains. Thus, there exist some truncation errors in its calculation by inter-sectoral cut-off and inter-regional cut-off effects (see Feng et al. 2011).

The top-down approach includes a mass balance analysis based on national statistics (e.g., Gu et al. 2013) and an economic **input–output (IO) analysis** (e.g., Hertwich and Peters 2009). An IO table describes the sale and purchase relationships between producing sectors and consuming sectors within an economy. Environmentally extended IO framework links data of direct emissions or resource demands for all economic sectors (environmental pressure data) with the monetary transactions between these sectors (intermediate demand) and enables an allocation of these emissions or demands to the consumption of product groups in corresponding sectors (final demand) based on an IO table (Minx et al. 2009). A direct-impact coefficient vector is determined by the environmental pressure data for each economic sector divided by the gross output of the sector. The direct-impact vector is then multiplied by the Leontief inverse, or the total requirements matrix, to yield a total-impact coefficient matrix. A total-impact coefficient of an economic sector shows both the direct and indirect environmental pressure rippling through the complex supply chains. An environmental footprint is calculated as a total-impact coefficient matrix multiplied by final demand (see Miller and Blair 2009 for details in calculation methods). The advantages of the top-down approach are comprehensiveness for its coverage and the IO analysis has capability for assessing the entire supply chains. It is commonly used in analyses on the national and supra-national level. Single-region input–output (SRIO) analysis can avoid the inter-sectoral cut-off effect, and multi-region input–output (MRIO) analysis can avoid both the inter-sectoral cut-off and inter-regional cut-off effects. However, these IO analyses are limited by the numbers of sectors in the IO database used for analyzing detailed consumption (Feng et al. 2011). Hybrid methods, combining the two approaches, can be taken, but it requires careful consideration of the frameworks to avoid double counting (Lenzen 2008; Lenzen and Crawford 2009; Strømman et al. 2009).

5.2.2 Methodological Development of Nitrogen Footprint

Five different methods have been proposed to quantify the NF: N-Calculator (Leach et al. 2012), N-Multi-region (Oita et al. 2016a), N-Input (Shindo and Yanagawa 2017), N-Output (described in the discussion of Eguchi and Hirano 2019), and the Coupled Human and Natural Systems (CHANS) model (Gu et al. 2013). The N-Calculator method calculates the per-capita NF of a country as a sum of food part (originally from fertilizer and biologically fixed nitrogen input) and energy part (NO_x emissions from fuel combustion) for goods, services, transportation, and housings. The N-Multi-region method determines the NFs of countries by MRIO analysis, estimating reactive nitrogen emissions (differentiating different nitrogen chemical species) accompanied with internationally and domestically traded commodities of all sectors of the economy. The N-Input method quantifies the food NF as the amount of nitrogen input through chemical fertilizers and biological nitrogen fixation, considering a target state's agri-food trade with the world. The N-Output method is for the food NF and it accounts for nitrogen loss during food production and consumption as a sum of surplus reactive nitrogen to a country and indirect nitrogen loss abroad using NUE of imported food and feed. The CHANS model is a country-level mass balance approach assessing all the nitrogen flows between subsystems (e.g., cropland, livestock, human, industry, and traffic).

Among the five methods, the N-Calculator method was the first developed and most widely used under the N-Print project (www.n-print.org). The N-Calculator method takes a bottom-up approach for the food NF and a hybrid approach (a bottom-up approach complemented by a top-down approach) for the energy NF, while the other methods take a top-down approach. In the first step of the N-Calculator method, a country average NF is calculated as follows, and in the second step, an individual's NF is determined as a country average NF multiplied by the ratio of the amount consumed by the individual to the country average amount of consumption. The food NF of the N-Calculator method consists of the food production NF and the food consumption NF. The food production NF is calculated as a sum of "protein nitrogen intake of each food item" multiplied by corresponding **virtual nitrogen factors (VNFs)**, which indicate the amount of nitrogen loss to the environment during production per unit of nitrogen consumed to show reactive nitrogen emissions intensity of the food item. The food consumption NF considers "protein nitrogen intake" multiplied by "one minus the denitrification ratio of wastewater treatment". The energy NF consists of a bottom-up approach part and a top-down approach part. The bottom-up approach part sums up each activity data related to NO_x emissions multiplied by emission factors. The top-down approach part subtracts the bottom-up approach part from the total estimation by SRIO analysis. The N-Calculator method for individual consumers is linked to a web-based NF tool of N-Calculator (www.n-print.org/YourNFootprint). The N-Calculator method is also expanded to institutions (e.g., Castner et al. 2017; Leach et al. 2013; a web-based tool at unhsimap.org), food labels (Leach et al. 2016), reactive nitrogen emissions neutrality (Leip et al. 2014), reactive nitrogen spatial intensity (NrSI, Liang et al. 2018), communities (Dukes et al. 2020), cities (e.g., Xia et al. 2020), and watershed (https://secure.cbf.org/site/SPageNavigator/bay_footprint.html).

To calculate the per-capita NFs of different countries/regions, the country/region-specific VNFs for the food part of the N-Calculator have been developed firstly for the United States (US), Europe, Austria, Tanzania, Japan, Taiwan, and Australia (summarized in a review paper by Shibata et al. 2017 with Tanzanian VNFs updated by Hutton et al. 2017), followed by China (Guo et al. 2017; Zhang et al. 2018; Oita et al. 2020), Egypt (Elrys et al. 2019), India (Oita et al. 2020), and Sub-Saharan African countries (Elrys et al. 2021). During the country/region-specific VNFs development, VNFs for more detailed food categories, such as rice, wheat, maize, eggs, lamb, and small ruminants, were estimated in addition to poultry, pork, beef, milk, seafood (including finfishes, crustaceans, and mollusks, both inland and marine), grains, starchy roots, legumes, and vegetables.

Within the same food category, seafood was investigated further by Oita et al. (2016b) for all seven seafood subcategories in FAOSTAT (an international database for food consumption provided by the Food and Agriculture Organization of the United Nations (FAO) at fao.org/faostat/en), with consideration of fed aquaculture, non-fed aquaculture, and wild catch to improve the original VNFs having only a single category for seafood. This first step to properly link aquaculture farms and fishing grounds to our food NF is introduced in the first part of the next section. Seafood is also an important option for our food choices. A case study of the application of the modified N-Calculator for seafood is shown in the second part of the next section. Differences in beef were shown by Liang et al. (2016) as a much less impact of grass-fed beef (VNF = 7.4) to grain-fed beef (25.2). Organically produced food VNFs were found to be comparable to corresponding conventional food VNFs, except for the VNF of organically produced beef being 124% higher (Cattell Noll et al. 2020).

Since food was found to be the most dominant component in those studies, the N-Input method and the N-Output method have been developed for better consideration of food trading and for better understanding of countries for the input or output of reactive nitrogen (if it is within a country or in main importing partners), keeping sensitivity to detailed food items. The CHANS model has been developed to overcome high data demand for calculating detailed VNFs including reactive nitrogen emissions for industrial products other than chemical products in addition to food and energy. The N-Multi-region method has been developed to identify countries of reactive nitrogen emissions for each chemical form at all upstream production stages to obtain a global picture and role of international trade. The application of this comprehensive global nitrogen model is introduced in the third part of the next section.

5.3 Application of the Nitrogen Footprint Models

5.3.1 Model Analysis of Nitrogen Footprint of Seafood

A new NF model, a feed-sensitive NF model for the N-Calculator method, has been developed to assess the impact of seafood in detail (Oita et al. 2016b). The VNFs for each category of seafood are basically calculated by ratios of utilization, recycling, emissions, and treatment at each stage of production to yield two sets of VNFs for the world average and for Japan, an important seafood consumer, with consideration of trade (Fig. 5.2). The feed-sensitive NF model examines the feeding steps in detail to consider fish-based feed for fed-aquacultured seafood, differentiating fed-aquacultured seafood, non-fed aquacultured seafood, and wild-caught seafood, whereas the original NF model for the N-Calculator only considers plant-based feed for seafood as the same way as livestock. Accompanying the feed-sensitive NF model, important parameters for calculating seafood VNFs were explored for the more accurate evaluation of the seafood NF using fisheries and aquaculture data by FAO (FishstatJ Release: 2.1.0.) and auxiliary data. The feed-sensitive NF model evaluates the 2010 Japanese food NF of all seafood at 1.55 kg N cap^{-1} year^{-1} and ca. 45% of that is for fed-aquacultured seafood (0.7 kg N cap^{-1} year^{-1}), whereas the original N-Calculator model computes the Japanese food NF of all seafood 3.71 kg N cap^{-1} year^{-1} and ca. 90% of that is for aquacultured seafood (3.36 kg N cap^{-1} year^{-1}). The identified key factors for assessing the seafood NF are the proportions of fed aquaculture and of plant protein in feed. The world average VNFs for different categories of seafood are provided as 0.2 (non-fed aquacultured and wild-caught), 4.8 (freshwater and diadromous fish), 3.9 (demersal fish), 3.4 (pelagic fish and other marine fish), and 8.2 (crustaceans) for 2010 (Fig. 5.3). The results inform that eating more non-fed aquacultured and wild-caught seafood and less fed-aquacultured shrimps and prawns could reduce nitrogen pollution due to our food consumption as effectively as choosing poultry (VNF of 6.0; Shibata et al. 2014) instead of beef (12.4; Shibata et al. 2014). From the ecological risk

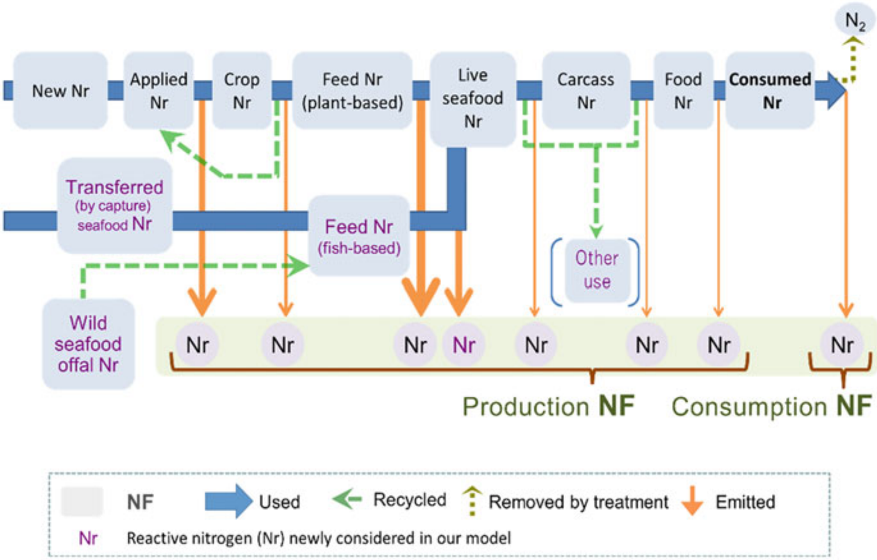

Fig. 5.2 A conceptual framework of the feed-sensitive nitrogen footprint (NF) model drawn from Oita et al. (2016b). The case of fed-aquacultured seafood is shown

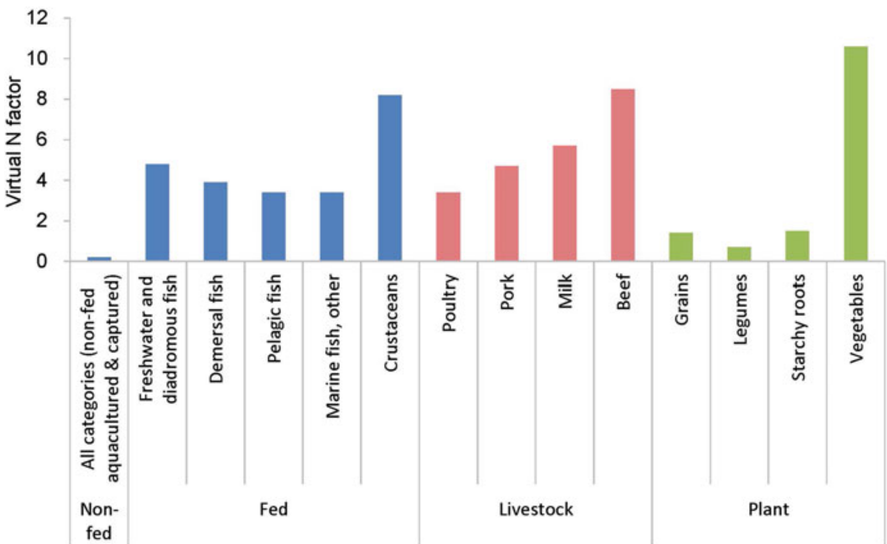

Fig. 5.3 Comparisons of virtual nitrogen factors (VNFs, nitrogen loss during production per unit of nitrogen intake). Seafood VNFs for the world average are taken from Oita et al. (2016b) and others are the common VNFs used for the United States and the Netherlands in Leach et al. (2012)

perspective, wild-caught seafood has a low food NF but its major problem for sustainability is overfishing. Sustainable fisheries including capturing species that have enough stock should be applied to minimize their overall environmental impacts. The feed-sensitive NF model gives a boost to integrated management of aquaculture, fisheries, crop farming, and livestock farming towards sustainable food production.

5.3.2 Historical Change of Japanese Food Nitrogen Footprint

The feed-sensitive model developed in the previous subsection was applied to Japan for the period 1961–2011 using VNFs of seafood for Japan by Oita et al. (2016b), VNFs of other food categories for Japan with consideration of trade by Shibata et al. (2014), and protein supply data by FAOSTAT (Fig. 5.4; Oita et al. 2018). Subsequently, four different diet scenarios were tested: (1) "Recommended level protein," reducing protein intake to the level recommended by the Ministry of Health, Labour and Welfare of Japan (Kido et al. 2012); (2) "Pescetarian," replacing meat protein intake with seafood; (3) "Low NF food," replacing protein intake from meat, dairy, eggs, and fed-aquacultured seafood with legumes and non-fed aquacultured and wild-caught seafood; (4) "Balanced Japanese diet," having a balanced diet as did in 1975 (traditional Japanese-based diet comprising Japanese Western cuisine), which

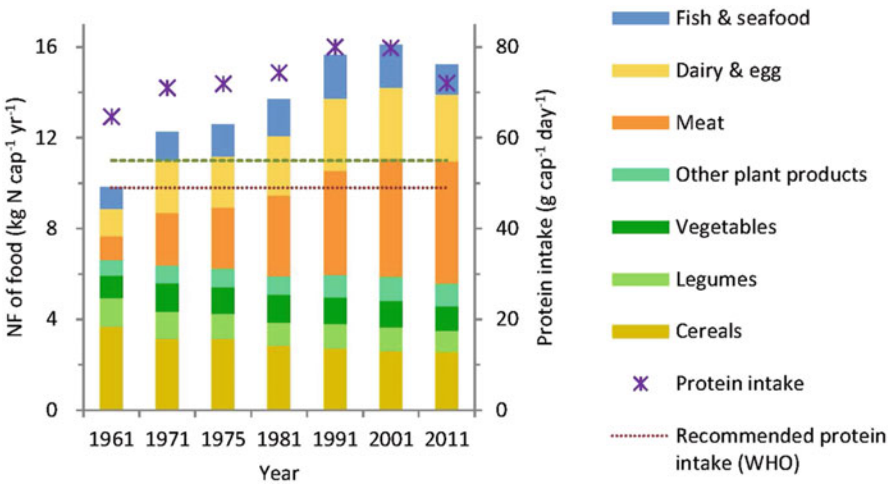

Fig. 5.4 Nitrogen footprint (NF) of food (kg N cap^{-1} $year^{-1}$) and protein intake (g cap^{-1} day^{-1}) in Japan and recommended levels of protein intake (g cap^{-1} day^{-1}) by the World Health Organization (WHO) and the Ministry of Health, Labour and Welfare of Japan (JMHLW). Values are taken from Oita et al. (2018)

is said to be well balanced in nutritional studies. Under all scenarios, the current food NF (as of 2011) will decrease by more than 15% (Recommended level protein scenario by 26%, Pescetarian scenario by 29%, Low NF food scenario by 45%, and Balanced Japanese diet scenario by 17%) (Fig. 5.4 for Balanced Japanese diet scenario). These results indicate that not only by reducing our protein intake but also by selecting low NF food to eat, we can lower our food NFs. The above findings are a step toward integrated risk management for sustainable food systems to deal with trade-offs between different kinds of issues including nitrogen pollution, stock depletion, human health, and carbon emissions.

5.3.3 *Model Analysis of the Global Nitrogen Footprint*

The global NF model for the N-Multi-Region method has been developed based on the environmentally extended MRIO analysis (Fig. 5.5). The global NF model determines the NFs of countries as the sum of direct and indirect emissions induced through their consumption, to air (emissions of NH_3, NO_x, and N_2O from all industries), and to water (nitrogen potentially exportable to water bodies N_{wp}, mostly emission potential of NO_3^- from agriculture and sewage). Emissions from agricultural sectors were estimated from fertilizer use by crop data (Heffer 2013) and FAOSTAT production and emissions data (at fao.org/faostat/en) using Intergovernmental Panel on Climate Change (IPCC)'s procedure (IPCC 2006). The estimated agricultural emissions and emissions data of all industries (Global Emissions EDGAR v4.2 FT2012) were linked to each of 15,000 industrial sectors in 188 countries/regions of a global MRIO table (Eora at http://worldmrio.com). Using the MRIO table extended with nitrogen emissions data, IO analysis evaluated the NFs for nations in 2010. The total global nitrogen emissions from agriculture and industries were 161 Tg N (75 Tg N of N_{wp}, 45 Tg N of NH_3, 35 Tg N of NO_x, and 6.2 Tg N of N_2O) and from consumers 28 Tg N. On a per-capita basis, the NF ranges greatly from under 7 kg N cap^{-1} year^{-1} in developing countries such as Papua New Guinea, Côte d'Ivoire, and Liberia to over 100 kg N cap^{-1} year^{-1} in wealthy nations such as Hong Kong and Luxembourg, with an average NF of 27 kg N cap^{-1} year^{-1} (Fig. 5.6). Looking at the emitters, 49% of the global reactive nitrogen emissions occur within the territories of four populous nations (China 20%,

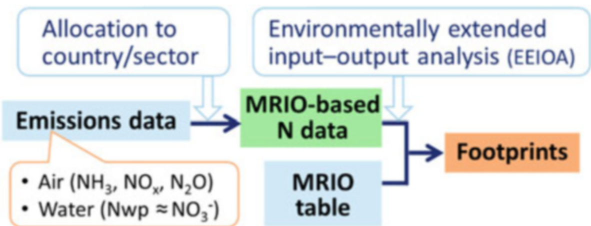

Fig. 5.5 Framework of the global nitrogen footprint model for the N-Multi-region method, drawn from Oita et al. (2016a). MRIO: Multi-region input–output

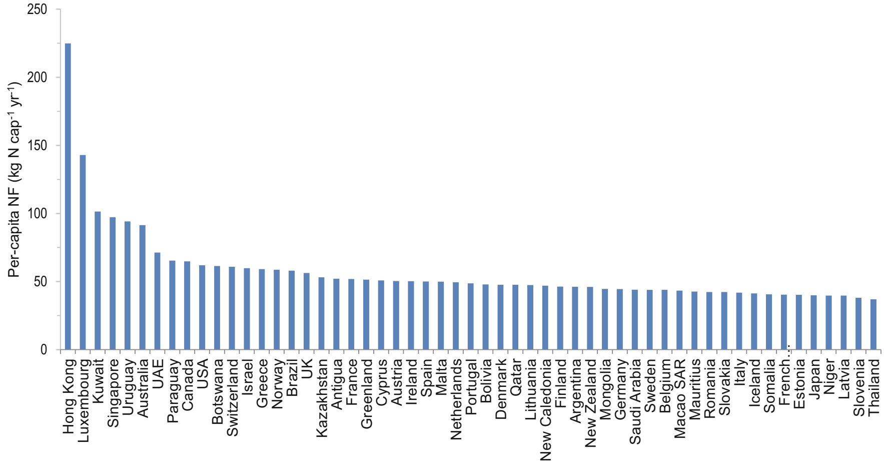

Fig. 5.6 The per-capita nitrogen footprints of nations for the highest 55 nations. Source: Oita et al. (2016a)

India 11%, the US 10%, and Brazil 6.1%), and an additional 12% occurs in another six nations (Russia, Pakistan, Indonesia, Australia, Mexico, and Argentina). Looking at the consumers, 46% of the global emissions are generated by production to meet the demand of the same four populous nations (China 19%, India 11%, the US 10%, and Brazil 6%), followed by another six nations (Japan, Russia, Indonesia, Germany, Mexico, and the UK) being responsible for additional 13%. The top emitters and the top consumers differ because 26% of the global NF is involved with the international trade of commodities. The main net exporters of the embodied reactive nitrogen emissions for products are nations bearing more domestic emissions for exporting products than emissions for importing products that occur in other countries. Those nations have a high capacity of agricultural, food, and textile exports, and are often developing countries. In contrast, the important net importers, causing more emissions abroad for importing products than emissions within their territory for exporting products, are mostly developed economies (Fig. 5.7). In terms of industrial sectors and major traded commodities, highly processed agricultural products (e.g., shirts, leather bags, and preserved food) play an important role in addition to meat and dairy products. The results reveal that international trade considerably drives local nitrogen issues such as NO_3^- or NO_2^- intake, air pollution, and freshwater pollution. Tracing complicated international supply chains has opened up new avenues in the understanding of anthropogenic nitrogen flows and the food-energy-water nexus.

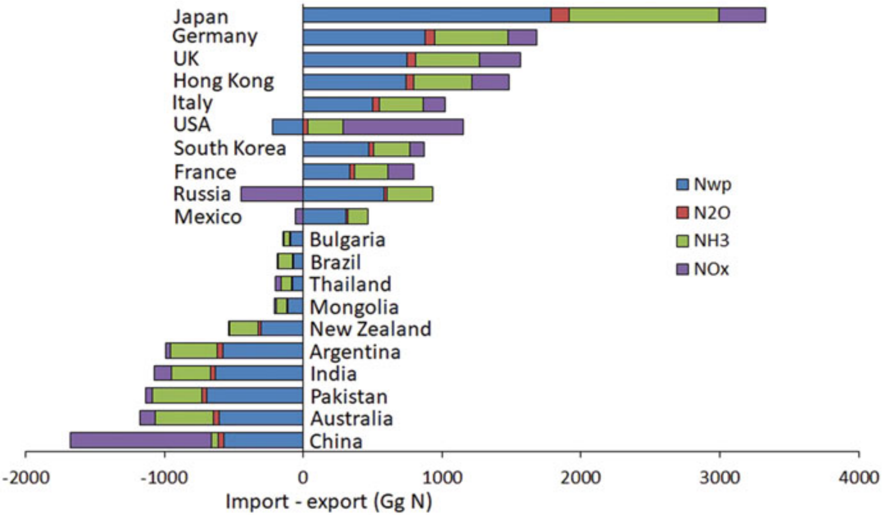

Fig. 5.7 Reactive nitrogen emissions embodied in traded products for the highest 10 net importers and exporters. The net importers are causing more emissions abroad by consuming imported goods than they bear emissions within the country for the production of exporting goods. Net exporters are bearing more emissions within the country for the production of exporting goods than they cause emissions abroad by consuming imported goods. Redrawn from Oita et al. (2016a)

5.4 Towards the Integrated Nitrogen Management

5.4.1 Ongoing Methodological Improvements and Applications of the Nitrogen Footprint Models

Further improvement of the NF models for increasing accuracy is needed since the NF models are all based on several assumptions and contain calculation errors. One issue is that the use of the VNFs only works for food that contains a sufficient amount of nitrogen. To improve the calculation accuracy of NFs of oil and sugar crops, Hayashi et al. (2020) proposed the use of VNFrees as a substitute for VNFs. The loss of reactive nitrogen calculated with input–output ratio and recycling ratio for each production process of a food item can be divided by consumed food weight itself to yield a VNFree, instead of being divided by nitrogen in protein content of consumed food to compute a VNF. Another issue is consumer behaviors. Consumer-level food waste was examined in detail by Hayashi et al. (2018). Some attempts looking at the difference in consumers include rural/urban lifestyles (e.g., by Cui et al. 2016) and food consumption of age and sex groups (Hayashi et al. 2018).

There is a growing need for integrating multiple footprint analyses to avoid risk trade-offs and to support sustainable development. Since nitrogen pollution is closely related to food production and water quality as basic human needs, the NF is an important indicator on measuring progress towards the Sustainable Development Goals (SDGs) by the United Nations (UN) (Vanham et al. 2019; San Martín 2020). In addition to previous attempts, e.g., with an LCA approach for the NF from a eutrophication perspective and CF (Xue and Landis 2010) and a grey water footprint approach for nitrogen and phosphorus (Liu et al. 2012), the NF model development provided combined food labels of NF, CF, and WF (Leach et al. 2016) and a combined institutional tool of NF and CF assessment introduced for several US universities (Leach et al. 2017). On a country level, the P-Calculator model was developed based on the development of the bottom-up NF models, to assess phosphorus footprint (PF) and NF in a common framework (Oita et al. 2020). The common assessment framework was applied to China, India, and Japan. The assessment for both the NF and the PF was expanded to Sub-Saharan African countries (Elrys et al. 2021) and the US (Metson et al. 2020). At a global level, further efforts include an expanded MRIO assessment of eutrophication impacts potential of nitrogen and phosphorus emissions to freshwater and marine water (Hamilton et al. 2018) and integrated environmental pressure assessment of greenhouse gas (GHG) emission, cropland use, freshwater use, and nitrogen and phosphorus application (Springmann et al. 2018).

5.4.2 Ongoing Efforts for Better Nitrogen Management

In efforts to address the global nitrogen challenge, one of the biggest issues is fragmentation across nitrogen science and policies (Sutton et al. 2019a, 2019b). Environmental policies related to nitrogen are usually separated by issue (climate, biodiversity, waste, etc.), by media (air, land, water, etc.), and by forms of reactive nitrogen (N_2O, NO_3^-, NH_3, NO_x, etc.) (Sutton et al. 2011). The stakeholders range from farmers and industries to conservation managers, policymakers, consumers, etc. In this regard, the NF provides overarching views in understanding nitrogen pollution in relation to our demand and its underlying lifestyles. Thus, the NF models serve as and are expanded to some of the key tools to be used for nitrogen management and for stakeholder's communication at a global level as well as that on a country level and at smaller levels. From the quantification point of view, the NFs are closely related to the quantification of nitrogen flows and impacts. It would promote further development of the NF models to work together with the recent efforts on assessing physical nitrogen inputs and outputs of systems or territories (called **nitrogen budgets**, e.g., Zhang et al. 2020) and NUEs along with supply chains and even longer full chains from production to consumers (Erisman et al. 2018; Uwizeye et al. 2016, 2020).

To address the fragmentation across nitrogen science and policies, the **International Nitrogen Management System (INMS)** was launched as a joint activity of the United Nations Environment Programme (UNEP) and the International Nitrogen Initiative (INI), an international network of nitrogen scientists. It is funded by the Global Environment Facility (GEF) Trust Fund and around 80 project partners through the “Targeted Research for improving understanding of the global nitrogen cycle towards the establishment of an International Nitrogen Management System,” or “Towards INMS,” project from October 2017 for 4 years (INMS at https://www.inms.international). This global scientific platform has activities consisted of four components: (1) Tools for understanding and managing the nitrogen cycle; (2) Quantification of nitrogen flows and positive and negative impacts; (3) Regional demonstration of tools; and (4) Awareness raising and knowledge sharing. These activities form part of a first comprehensive International Nitrogen Assessment (planned to be published in 2022). Some of the activities are working on detailed guidance documents on assessment methodologies and measures for better nitrogen management including the NF models and applications (Sutton et al. 2019a).

Efforts in policy frameworks to integrate the global nitrogen challenge to ecological risk management and other environmental risk management realized the UN Resolution on Sustainable Nitrogen Management as a key milestone (UNEP 2019). The INMS is working with the UNEP to establish an Inter-convention Nitrogen Coordination Mechanism (INCOM) (Sutton et al. 2019a). Through UNEP and with scientific support by the INMS, this new institutional infrastructure is intended to bring together the Inter-governmental Conventions and Programmes including the UN Convention on Biological Diversity (CBD), the UN Framework Convention on Climate Change (UNFCCC), the Global Programme of Action for the Protection of the Marine Environment from Land-based Activities (GPA), the UNECE Convention on Long-Range Transboundary Air Pollution (LRTAP), and the Vienna Convention for the Protection of the Ozone Layer. In parallel, based on the current NF and related studies, further research would need to fill the knowledge gap about detailed relationships between our demand as drivers, nitrogen pollution as causes, and effects on ecosystems.

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Chapter 6
Adaptive Risk Management of New Coronavirus Disease

Hiroyuki Matsuda and Akira Watanabe

Abstract The new coronavirus (COVID19) is an infectious disease that has a relatively high infectivity and high mortality rate. To make matters worse, it is infectious even before the onset, and it is difficult to take measures against infection. The emergency measures taken by the Japanese government in the spring of 2020, requiring the general public to self-isolate where possible, succeeded in drastically reducing the number of people infected. The economic loss, however, was estimated to be in the vicinity of 10 billion yen a day in Tokyo. It is well-acknowledged that there is a trade-off between maintaining economic activity and the prevention of an outbreak of disease. The Susceptibility-Infected-Recovered (S-I-R) model has been used to estimate the effect of the intervention on the number of fatalities due to infection. The labor loss due to isolation and the relationship between the number of deaths by the infection and economic loss are calculated as a function of the degree of intervention. The economic loss is considered to be a concave increasing function of the degree of intervention. We assumed that the situation will be resolved medically within 2 years. When the basic reproduction number (R0) is high, it is difficult to halve the number of fatalities without increasing the intervention coefficient. As such, it is necessary to choose either a strategy of suppression or one of mitigation. In the case of smaller R0, the number of fatalities can be reduced to some extent by increasing the degree of intervention.

Keywords S-I-R model · Non-pharmacological intervention · Herd immunity · Suppression · COVID19 · PCR testing

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6.1 Mathematical Model for Epidemics

In 2020, the world faced the coronavirus pandemic. The city was first closed in Wuhan, China on January 23, 2020. The next mass infection was discovered on a British-registered cruise ship Diamond Princess anchored at Yokohama Port. The ship entered long-term quarantine on February 4. The infection has spread in Europe and the USA in March, and was recognized as a pandemic on March 11. Infectious diseases are also an important issue in ecological risk management.

Epidemic models are very similar to those of population ecology. The main difference is that it deals with the number of infected, noninfected, and recovered individuals (number of people who have acquired immunity), instead of the number of pathogens. The following are the famous SIR models (Kermack et al. 1927, Handel et al. 2007).

Let S, I, R mean the number of susceptible, infected, and recovered people, respectively. We assume that the total number $N = S + I + R$ is constant.

$$\frac{dS}{dt} = -(1-f)\beta SI \tag{6.1}$$

$$\frac{dI}{dt} = (1-f)\beta SI - \gamma I$$

$$\frac{dR}{dt} = \gamma I$$

where β, γ, and f are transmission rate, recovery rate, and the reduction in transmission due to intervention strategies. Hereafter we call f the intervention coefficient.

There are two types of intervention measures, pharmaceutical and non-pharmaceutical. Pharmaceutical intervention means a recommendation initiated by a pharmacist in response to a drug-related problem in an individual patient occurring in any phase of the medication process such as vaccination and antivirus medicine (Ferguson et al. 2020). Non-pharmaceutical intervention (NPIs) means public health measures aimed at reducing contact rates in the population and thereby reducing transmission of the virus (Ferguson et al. 2020).

The quantity $\beta N/\gamma$ is usually referred to the basic reproduction number R_0. The basic reproduction number is the expected number of secondary infections produced by an infected person when everyone is susceptible. If $R_0 > 1$, the infected number will increase when $S = N$, i.e., $dI/dt > 0$.

There is a threshold of S, denoted by S_{th}, below which the infection does not increase, i.e., $dI/dt < 0$. The threshold is given by:

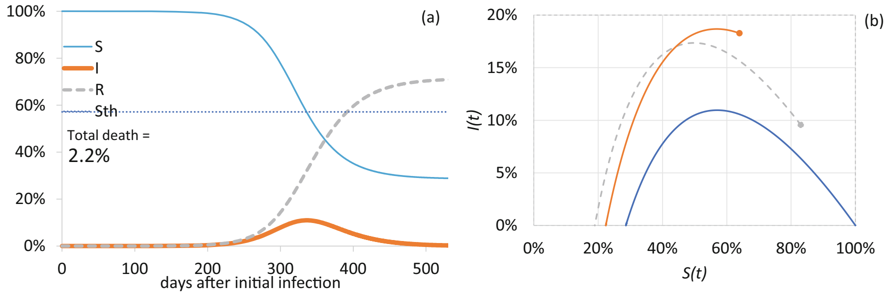

Fig. 6.1 Dynamics of susceptible, infected, and recovered population of the SIR model (6.1). (**a**) infectious and recovered increased with decreasing susceptible people. If $S(t)$ is below S_{th} (58%), I (t) began to decrease. Parameter values are $(\beta N,\ \gamma,\ f) = (0.125,\ 0.05,\ 0.3)$, $(N,\ I(0),\ R(0)) = (1{,}1000{,}000,\ 30,\ 0)$. (**b**) The phase diagram of $S(t)$ and $I(t)$, where $(S(0), I(0)) = (N - 30, 30)$ (bold), $(83\%\ N, 10\%\ N)$ (broken), and $(64\%\ N, 18\%\ N)$ (dotted), respectively

$$S_{th} = \frac{\gamma}{\beta(1-f)} \tag{6.2}$$

S_{th}/N is the inverse of R_0 if $f = 0$. If $R_0 > 1$, the infected number will increase until the susceptible people decrease below the threshold S_{th}. If $R_0 = 2.4$. Figure 6.1a shows a simulation result of $I(t)/N$ and $R(t)/N$ which shows an outbreak.

The SIR model has no positive equilibrium, implying that infectious disease temporally outbreaks until the susceptible people decrease below S_{th}, and finally disappear as the population gets the "herd immunity" or "population immunity," which means a form of protection from infectious disease when a population has become immune to an infection (Handel et al. 2007). The final percentage of susceptible people, $S(\infty)/N$, depends on the initial state $(I(0), R(0))$, as shown in Fig. 6.1b. $(83\%\ N, 10\%\ N)$ and $(64\%\ N, 18\%\ N)$ mean states of 153 and 168 days without intervention. Delaying the intervention does not always increase the cumulative infected people at the time of convergence.

R_0 depends on the intervention coefficient f. The timing and the maximum ratio of infected people are delayed and lower as the intervention coefficient increases, as can be seen by comparing Fig. 6.1a ($f = 0.3$) and Fig. 6.2a ($f = 0$).

There are two fundamental strategies for measures to infectious disease, mitigation, and suppression (Ferguson et al. 2020). Mitigation strategy does not stop epidemic spread but tries to reduce peak healthcare demand and obtain herd immunity. Suppression strategy aims to prevent the outbreak and reduce case numbers to low levels. "Optimal mitigation policies" combine "home isolation of suspect cases, home quarantine of those living in the same household as suspect cases, and social distancing of the elderly and others at most risk of severe disease" (Ferguson et al. 2020), which might reduce peak healthcare demand and deaths.

The herd immunity is obtained under mitigation strategy if $f < 1 - \beta N/\gamma$. The outbreak is suppressed if $f > 1 - \beta/\gamma$, as shown in Fig. 6.2 ($f = 0.6$). In the case of

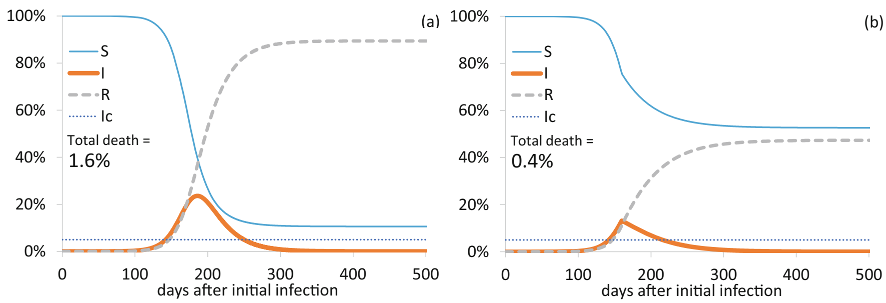

Fig. 6.2 Dynamics of susceptible, infected, and recovered population of the SIR model (6.1). (**a**) For the case when $f = 0.0$, $I_c = 5\%$, and $I(0) = 30$ (mitigation strategy), a small outbreak happens and the cumulative ratio of infected people is 41% and the total mortality is 1.6%. (**b**) For the case when f changed from 0 to 60% at $t = 160$ (suppression strategy), an outbreak is suppressed because $R_0 \leq 1$ for $t > 70$. The total mortality was 0.4%

suppression strategy, the cumulative ratio of infected people strongly depends on the ratio of infected people at the beginning of intervention measures.

The ratio of cumulative infected people is obtained by $1 - R(\infty)/N$ because all infected people recover or die in the SIR model. We interpret that R includes the fatal case in the SIR model. The case fatality risk (CFR), the mortality ratio of confirmed case patient, depends on the age of the patient. CFR is 0.2% for 40 years old or younger case and 14.8% for 80 years old or older case (Nakazawa M., pers.com[1]). We note that the infection fatality risk (IFR), the mortality ratio per infected person, is lower than CFR because some infected persons recover spontaneously without confirmation. The overall IFR depends on the age structure and medical equipment of the country. Here, we assume the overall IFR in Japan is 1.6% (Nakazawa M., unpub[2]).

We assume that the total number of deaths before the epidemic subsides, denoted by D, is given by the following expression:

$$D = \int_0^T d\gamma[I(t) - I_c]dt \tag{6.3}$$

where d is the IFR per day; I_c is the number of beds that can secure patients; T is the period until the pandemic ends due to the development and spread of the vaccine, and is assumed to be 2 years. For simplicity, we assume no patient dies if $I(t) < I_c$. Otherwise, the infected patient will recover alive or die with probability 1-IFR:IFR, where IFR is 1.6%. Therefore, we assume that d=1.6%. We also assume that $I_c/N = 1\%$ or 10%.

[1]Nakazawa M. http://minato.sip21c.org/COVID-19.pdf, accessed 20 Jul 2020.

[2]Nakazawa M. http://minato.sip21c.org/COVID-19.pdf.

6.2 Trade-off Between Infectious Mortality and Economic Impact

There is a trade-off between mortality risk and economic cost of intervention measures. The total number of deaths before the epidemic subsides will be the cumulative number of people multiplied by the IFR. D depends on the intervention coefficient f. The higher the f is the smaller the D. However, the higher the f, the higher the economic loss. The economic loss due to emergency measures from April 8 to May in Japan is estimated to be 140 billion US$ per month (Takahide Kiuchi, unpub[3]).

The former is not economic cost, but some method exists the value of statistical (or probabilistic) life (VSL). VSL is defined as the inverse of the ratio of the risk of a given death to the amount willingness-to-pay (WTP) to avoid it. We assume that VSL is 2 million US$ per person, although Thunström et al. (2020) estimated 10 million US$. Namely, we may accept 0.01% mortality to get 200 US$.

We assume that the economic loss due to intervention measures, denoted by C, is $C = C_0(f/f_0)^q T_1$, where C_0 is the economic loss per day when $f = f_0$; f_0 is the f when the emergency period during April 2020 in Japan; q is the magnitude of nonlinearity and T_1 is the period until $S(t)$ reaches S_{th}, or the period to acquire herd immunity. We assume that f becomes 0 after $S(t) < S_{th}$ because the number of infected people will decrease even without intervention after the herd immunity is acquired. We consider that C can increase disproportionately with f because there are various means of interventions that have different efficiencies between measures, as shown in Ferguson et al. (2020). Although it is possible to reach a small f by effective intervention measures, it would result in a huge economic loss to achieve a high f as in an emergency. Hereafter we assume that q is 1, but more accurate analysis will be needed based on the data of each country. In the case of Japan, the economic loss during the emergency measures from April 8 to May 20 was estimated to be 139 billion US$ per month (Kiuchi 2020, https://www.nri.com/jp/knowledge/blog/lst/2020/fis/kiuchi/0430_3), where we assumed that 100 yen = 1 US$. If we assume that $q = 1$ and $f_0 = 0.6$ during this period, $C_1 = 4.6$ billion US$/day. T_1 depends on the intervention effort.

We consider two scenarios; scenario I for $(\beta N, \gamma) = (0.07, 0.05)$ and II for (0.125, 0.05), respectively. The basic reproduction number $R_0 = 1.5$ and $R_0 = 2.4$ without intervention for scenarios I and II, respectively. Figure 6.3a–d show the impact of the intervention on the economic loss, loss of VSL, and the length of intervention period for scenario I and II, respectively. We assume that the intervention will end when the herd immunity is acquired or $S(T_1) = S_{th}$, although new infections will not end but decrease. In Fig. 6.3, the length of the intervention period (T_I) becomes longer as f increases.

We consider I_c/N is small (1%) and higher case for both scenarios. The suppression strategy is implemented if $f > 1-1/R_0$, or if $f > 39\%$ and 59% for scenarios

[3]Kiuchi T. https://www.nri.com/jp/knowledge/blog/lst/2020/fis/kiuchi/0430_3, accessed 20 Jul 2020.

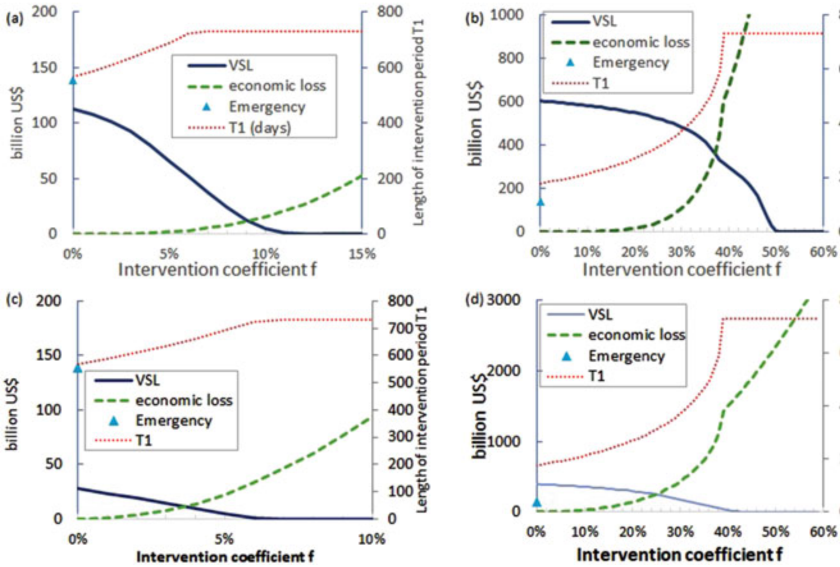

Fig. 6.3 The economic loss due to intervention and infection mortality in the case when (βN, I_c/N, q) is (**a**) (0.41, 1%, 2), (**b**) (0.61, 10%, 2), (**c**) (0.61, 1%, 4), and (**d**) (0.61, 5%, 4). Triangle on the vertical axis means the economic loss of emergency during April 2020

I and II, respectively. However, if $f > 23\%$ and $f > 39\%$ for scenarios I and II, respectively, the pandemic will end (T is 2 years we assumed) due to vaccination or other medicines before the herd immunity is acquired, namely $S(T) > S_{th}$.

In the suppression strategy $C = 133$ and 306 trillion US$ for cases (a) and (b), respectively, because the intervention must continue until the end of the pandemic. Although the functional form of the economic loss due to intervention C is uncertain, the total reduction without intervention depends only on the number of deaths, whose costs are estimated by the number of cumulative deaths and VSL per person. The number of cumulative deaths were 789,000 and 440,000 for case (a) and (b), respectively. This depends on I_c If $I_c = 0$ (no beds to save critical patients), the number of cumulative deaths are $N(1 - 1/R_0)$, which means 75,000 and 113,000 for scenarios I and II, respectively.

In scenario II, it takes about half a year if $f = 0$ until the acquisition of herd immunity. If $f > 39\%$, herd immunity cannot be obtained for 2 years. If $f > 51\%$, there will be no medical collapse in which economic loss is 316 billion US$. When $q = 4$ and f ranges from 0 to 35%, VSL ranges from 60 to 40 billion US$, and the economic loss ranges from 0 to 25 billion US$. This has resulted from medical collapse. Case (d) is not very different from the result of case (c), but if $f > 43\%$, medical collapse will not occur for 2 years. If I_c/N is less than 8–10%, medical collapse cannot be avoided if a mitigation strategy is adopted. At $q = 4$ and $f = 28\%$, VSL ranges from 40 to 11 billion US$ and economic loss ranges from 0 to 25 billion US$.

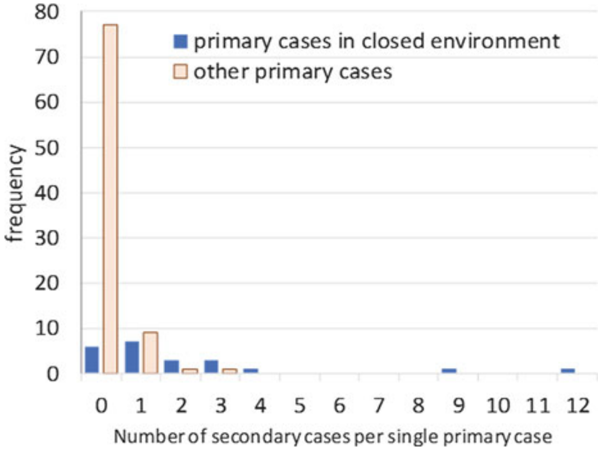

Fig. 6.4 The distribution of the number of secondary cases generated by a single 93 primary cases with CoVID-19 among 110 cases in Japan (Nishiura et al. 2020)

6.3 Adaptive Management for CoVID-19

CoVID-19 infection has been found that many infected people affect zero or very few people, but only a small proportion, called "super-spreaders," have infected many susceptible individuals, as shown in Fig. 6.4. It is important to identify the people who have come into close contact with the super-spreaders and to prevent further transmission by following the path from who to whom and when. A population of several to several tens of patients whose infection routes are being followed is called a "cluster." The figure shows the results of a survey of a total of 110 infected cases among 11 clusters infected in Japan as of February 19, 2020. In some spreaders infected 12, 9, and 4 people, but more than 80 cases did not infect any others. "All clusters were associated with close contact in indoor environments, including fitness gyms, a restaurant boat on a river, hospitals, and a snow festival where there were eating spaces in tents with minimal ventilation rate" (Nishiura et al. 2020). It is important to prevent secondary infections from super-spreaders, as sporadic infections do not spread.

Adaptive management is often done in very poor information. This is also the case with the nuclear accident risk assessment introduced in Chap. 3. In the fishery management described in Chap. 7, there are large uncertainties in recruitment rates, and there are also large errors in the estimation of stock abundance. Regarding the new coronavirus countermeasures, the basic production number, the effect of the intervention measures, and the number of newly infected cases were uncertain, and the countermeasures were being advanced. The economic loss calculated in the previous section is something that can be known after an emergency is actually taken, and it was not known whether the number of newly infected cases could actually be contained. It is easy to criticize afterwards that, for example, such extreme measures were unnecessary. When declaring an emergency, what was important was to ensure suppression. It is valuable that the success of suppression was once achieved. Initially, we used data for parameter values of new coronavirus from foreign countries such as Wuhan in China, but it is important to update the information shown in Fig. 6.3 while taking measures in Japan.

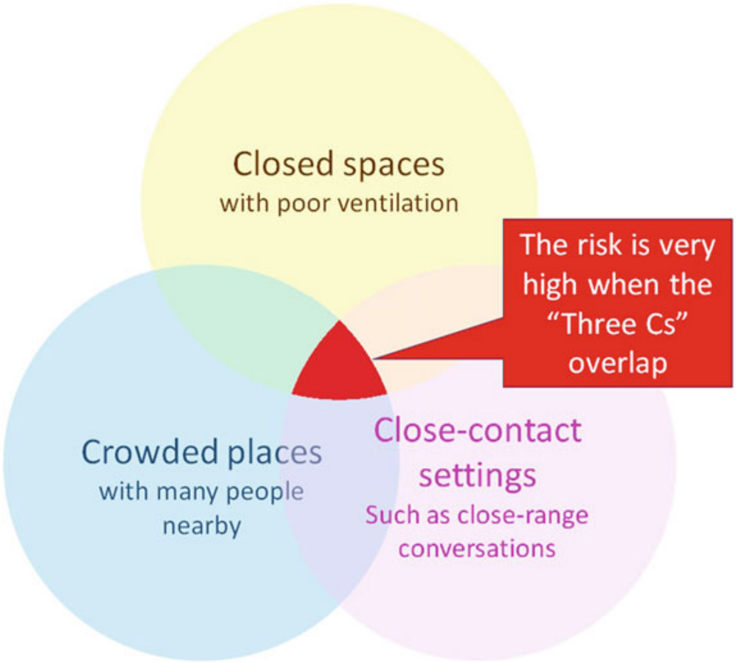

Fig. 6.5 Key summary from the COVID-19 countermeasures experts meeting (https://www.mhlw.go.jp/content/3CS.pdf, accessed 21 Feb 2021)

The key to preventing secondary infection is finding more clusters as soon as possible. To do so, it is important to protect the private information of infected people to maintain an honest reporting situation and to detect areas where secondary infections may occur. When the number of infected people with unknown sources increases, behavioral change is required not only for clusters but also for society.

The mean and standard deviation were 0.6 cases and 1.9 cases, respectively. The R_0 among primary cases in closed environment and among the other primary cases are 2.1 and 2.9 cases, respectively. It is said that "civil behavior change" is effective to decrease the infection rate. As shown in Fig. 6.1, people gather in a closed room with poor ventilation (Closed spaces), engage in dense contact, especially in conversations that may cause droplet infection (Closed contact), and crowds within reach. (Crowded places) "three 'C'" condition is said to be dangerous. It is said that it is important to keep a social distance, such as not having face-to-face conversations, and being careful to keep coughs away from passing each other (Fig. 6.5).

In addition, as mentioned in Sect. 6.1, early diagnosis of patients and enhancement of intensive care for severely ill patients and securing a medical provision system reduced the number of deaths, earned time until the collapse of medical care, and increased the freedom of policy. Is extremely important in increasing When beds for severely ill patients become scarce, "triage" is needed for those with mild illness to recommend home medical treatment. There is a risk that all people will not be able to take all possible measures. There may be situations where it is necessary to seek not individual optimization but group optimization.

The key to preventing secondary infections is to find clusters as soon as possible. To that end, maintaining honest reporting status of who the confirmed infected person has contacted and protecting the personal information of infected people in order to detect potential areas of secondary infection. As the number of infected people of unknown origin increases, behavioral changes not only in the clusters but also in society are required.

It is said that "behavioral change of citizens" is effective in reducing the infection rate. As shown in Fig. 6.4, people gather in a closed room (closed space) with poor ventilation, especially in conversations that can cause droplet infection (closed contact), and are within crowded areas. It is said that the "3-C" state is dangerous, but it is important to maintain a social distancing such as face-to-face conversation and taking care not to cough.

The average and standard deviation of the number of effective reproductions estimated from the frequency distribution in Fig. 6.4 were 0.6 and 1.9 cases, respectively. In other words, if this distribution reflects the population, infectious diseases should end in Japan without an outbreak. However, if there are 11 out of 110 samples was super-spreaders satisfying the three C conditions, but the true proportion of super-spreader in the population is higher, $R_0 > 1$. In Japan, after March 20, the number of newly confirmed infected people increased rapidly, and it became an emergency declaration on April 8.

Suppression may be possible if proper measures are taken against super-spreaders. This may differ from what occurred in Europe and the Americas, but R_0 can vary from country to country.

We implicitly assumed that the quantity of $\beta N/\gamma$ is known and f can be manipulated accurately by the policy. However, the quantities of $\beta N/\gamma$ and f under specific behavioral suppression are usually estimated after the end of the pandemic. As will be explained in detail in a later chapter, new recruitment rates of deer do not change drastically every year, unlike fisheries resources and bears changes their recruitment rates. Assuming that the infection interval x, which follows the Weibull distribution $\Pr[X < x] = 1 - \exp[1 - (x/b)^a]$. It is estimated that a and b of the Weibull distribution were estimated to be 2.305 and 5.452, respectively, from the information on the number of new infections from January 3 to May 10, 2020. That is, the median infection interval is estimated to be 4.65 days and the 95th percentile is 8.78 days. This statistical estimation method and the used data[4] were explained and discussed in the public "webinar"[5] on May 12. The data is released almost every day. The decision-making is so frequent, unlike fisheries and wildlife management is usually revised every year. As with fisheries management, adaptive management techniques are useful.

Monitoring is indispensable for adaptive management. The data updated daily is the numbers of PCR tests and those who test positive. At least in Japan, it is unsure how much the number of those who test positive reflects the increase or decrease in

[4]https://github.com/contactmodel/COVID19-Japan-Reff/tree/master/data, accessed 20 Jul 2020.

[5]https://live2.nicovideo.jp/watch/lv325833316, accessed 20 Jul 2020.

the true number of new infections in the population. In addition, 80% of the behavior regression was required during the emergency, and the behavioral suppression rate of the Japanese is far below 80% by big data analysis. It was reported that behavior did not reduce by far smaller than 80% (Apple https://www.apple.com/covid19/mobility, accessed 9 Jul 2020). However, the behavioral suppression does not mean the intervention coefficient, it is unknown how much the actual f is suppressed. What is important is to keep the number of effective reproductions to less than 1, not to reduce the behavior suppression rate to 80%. Controlling 80% of behaviors is a means to increase f and reduce the reproduction number, not the purpose.

In addition, there were criticisms that the number of PCR tests was too few and nonrandom sampling in Japan. Thus, the number of confirmed infected people is not the "true number of infected people" for the entire population in Japan. In that case, we estimate the increase/decrease trend of the true number of cases by adjusting sampling bias. It is known that PCR tests for new coronavirus include false negatives therefore the multiple tests are needed to confirm true negative. If the test is not aimed at finding out who is positive, then a 100% or thorough investigation is not necessary. For adaptive disease control, we need to monitor the relative value of the true number of infected cases based on the limited and biased test samples.

The basic reproduction number (R_0) is estimated to be 2.5, but this value also has uncertainty. Recognizing this uncertainty, we will create a mathematical model and devise countermeasures. The effectiveness of the measures may be evaluated by the cumulative number of cases. If the effects are insufficient, the measures may be strengthened, and if they are effective, the measures may be relaxed or tightened to the extent that the number of newly infected persons can be reduced or increased. This is "adaptive management" in population ecology.

The PCR test in Japan is not a method of knowing the true number of cases, but a method of detecting second infected persons from those who have had close contact with the infected person and suppressing further infections. Therefore, the sample of persons tested does not reflect the entire population, age and location may be biased. Moreover, the magnitude of bias may vary from week to week.

We never recommend to abandon measures to prevent secondary infections and use PCR tests to estimate the true number of people infected. However, it is necessary for adaptive management to know the trend of the number of new infections.

Since the inspectors are not a random sample of the population, the whole picture cannot be seen from the rate of test positive. The number of fatalities is helpful to estimate the true number of deaths due to new coronavirus, but it takes about 2–3 weeks from infection to death. For adaptive management, the time-lag to monitor the state is longer, and the change in measures will be delayed. It is possible that the increase or decrease in the true number of infected persons can be estimated to some extent by comparing the gap in attitude distributions between people tested and the entire population such as age, sex, occupation, location, etc. Then, the effect of measures such as behavior suppression can be verified within several days. As of June, estimation of the effective reproduction number has a time-lag of 2 weeks. Two digits estimation is not possible, but a provisional value should be available in about 9 days if the period from the first infection to the second is 9 days at the 95th percentile. What is important is not to improve the accuracy, but to respond promptly with a management system that does not make a mistake in policy.

In Japan, in addition to preventing medical collapse, the key is to maintain a system where "Cluster Response Team" tracks the infection route of infected persons and prevents chain of infection. The emergency declaration was issued on April 8 as the number of new infections increased and the number of people with unknown infection routes increased and exceeded the capabilities of this team. Therefore, if the number of new infections was set to "a level that can suppress the spread of infection," the emergency situation can be lifted. However, the current evaluation method and evaluation criteria for policy change necessary for adaptive management have not yet been established.

There is another idea to increase the number of staff in the cluster response team is important. On June 20, an app that can detect people who have contacted infected people with smartphones was released in Japan. This can help to detect close contacts and strengthen the cluster response team.

In Japan and other countries, we have once succeeded in reducing the number of new infections by regulation of citizens' behavior. While carefully monitoring the increase/decrease in the number of newly infected cases with a delay of about 10 days, how much behavior regulation can reduce the intervention effort while keeping the effective reproduction number is less than 1.

The mathematical model described here is not realistic because it is simple and does not consider the incubation period, effects of immigrants, and the cluster response team. However, this theory has shown two fundamental strategies of mitigation and suppression.

Even if the value of R_0 and the relationship between f and behavioral regulation are uncertain, management will be successful if the number of new infections continues to be reduced. If the number of new infections increases, we need to review the management policy. This is called "feedback control." It is the same as if you do not know the horsepower of your car, you can drive by monitoring the speedometer. Then, while implementing countermeasures and verifying the effects, it is adaptive management in ecology that the true value of R_0 can be known by using Bayesian estimation.

The mathematical model is not the reality. Even if the behavior is suppressed by 80%, R_0 does not always decrease by 80%. Even if the number of contacts is the same, the quality is quite different if you frequently meet with your family, meet unknown people in the city, and move further to meet people outside the city. After all, it is important to estimate the true number of new infections and verify the effect by monitoring the effective reproduction number. The intention of 80% control is that measures to ensure a reduction were necessary in an emergency because we do not know the exact parameter values of R_0 and f.

When ending an emergency, it is also important to consider which behavior regulation should be mitigated. According to the 9th Report from the Imperial College of London, school closure has only a 2–3% effect on curbing deaths. In this way, it is possible to estimate the effects and propose areas where the regulation is canceled, but to treat the areas where the suppression is continued, and gradually recover daily life while preventing newly infected persons from increasing again.

There were reports and opinions that confuse purposes and measures. Not only the response team but also the media and people who change their behavior are required to take strategic measures.

Finally, it was reported that there were some people in Japan who did not lock down cities with penalties and some people went outside even during the emergency period. Many foreign people asked the reason. From the reflection on the pre-World War II system, the post-war Japanese system places the highest priority on respect and security of the people's freedom. Therefore, the whole legal system is designed to avoid the government from restricting the freedom of the people by order. Many other countries have more nation-based penalized orders and instructions. Instead of guaranteeing freedom, each citizen must take responsibility for his or her own judgment.

When working with public officers, it is often seen that bureaucrats are very cautious about restricting the freedom and rights of the public, and individuals are free to oppose any proposal of these restrictions. As far as the new coronavirus epidemic is concerned, we can say that the Japanese system is working well so far.

In conclusion, suppression strategy needs forever intervention until vaccination is developed or the virus disappears from the world. Suppression strategy is possible irrespective of the infection rate ($f > 1 - \gamma/\beta N$). Mitigation policy may stop further explosion if $S > S_{th,\, f=0} = \gamma/\beta N$. Either the mitigation or suppression strategy, the number of beds should be as many as possible if the cost is smaller than the economic damage.

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Part II
Fisheries Risk Management

Chapter 7
How to Convince Purse Seiners for Sustainable Fishery

Hiroyuki Matsuda and Hiroaki Kawai

Abstract The capture fishery is characterized by the fact that the stock-recruitment fluctuates greatly depending on the environment such as water temperature, and the estimation of stock abundance and other life-history parameters such as natural mortality rate are highly uncertain. These are why risk management is essential. It is difficult to convince fishers to understand that resources will increase if overfishing is avoided. Previously, fishery management was proposed based on the "maximum sustainable yield" theory, which ignores uncertainty and assumes to reach a steady-state. Recently, adaptive management theory that considers measurement error and process error has been developed, and risk management has been well-established. Bayesian inference is useful for estimating stocks from catch data. This estimation method is suitable for adaptive management that manages resources while updating information and unverified assumptions. We need to take care when detecting the density effects of the spawning stock biomass on the recruitment from time-series data. Especially for marine resources, the annual process error may not be independent, and there is a decadal change in the system in which the sea surface temperature of the spawning ground. Such decadal change in environmental conditions might lead to strange interpretations of stock abundance estimation. This chapter introduces the theory of adaptive resource management from the history of persuading Japanese purse seine fishers to manage the resources of chub mackerel.

Keywords Adaptive management · Measurement error · Process error · Overfishing · Stock–recruitment relationship · Total allowable catch · Maximum sustainable yield

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7.1 The Classic Theory of Maximum Sustainable Yield (MSY)

Risk assessment is incorporated into fisheries management as well as standards for environmental chemicals. Use of marine resources definitely affects the target organisms. Since overfishing impairs future profits, it is desirable to use it sustainably. However, the rate of stock increase and the amount of fishery resources are uncertain and fluctuate yearly. Therefore, management rules based on risk management will be applied.

Since the United Nations Convention on the Law of the Sea came into force in 1996, each member state has the responsibility to use the fisheries resources in a sustainable way, as well as having the rights to use them exclusively. More specifically, the total allowable catch (TAC) is to be determined as a threshold of overfishing. At present, Japan has set TACs for seven fish species such as sardine (*Sardinops melanostictus*), Pacific saury (*Cololabis saira*), walleye pollock (*Gadus chalcogrammus*), and snow crab (*Chionoecetes opilio*), and uses all fisheries resources in the Japanese EEZ exclusively (Matsuda et al. 2010).

The capture fisheries capture the surplus resources while leaving the parent that reproduce moderately. Even if we do not catch it at all or get all, it is unable to continue to use the resource effectively. This is similar to the interest life from saving bank accounts. The amount of interest is determined by the amount of deposit and the interest rate. If you withdraw more than interest, your deposit will decrease and your next year's interest will decrease. In order not to reduce the amount of deposit, you have to withdraw not more than the amount of money than interest. It is better not to catch any fish at all if we want to reduce any negative effects. The whaling controversy is, after all, a conflict between the values of the argument "avoid any risks" and the argument "reasonably manage risk for sustainable whaling." Fishing is a risk-prone industry and uses natural resource (Matsuda et al. 2015).

There are four major differences between capture fisheries and savings bank accounts. (1) the rate of stock increase depends on the stock abundance, while the interest rates do not vary by principal. As shown in Fig. 7.1, as the number of deposits increases, interest rates decrease. For biological resources, the growth rate slows down as the population increases, and eventually stops growing. In ecology, this is called the "density effect." The maximum population size of the species that the environment can sustain indefinitely, given the food, habitat, water, and other necessities available in the environment is called "carrying capacity" (Fig. 7.1).

Let N_t be the stock biomass in year *t*. $N_{t+1}–N_t$ is called the surplus productivity, which would correspond with the interest of bank saving. It is usually considered that the surplus productivity is a one peak curve with respect to stock biomass N_t, as shown in Fig. 7.1. The stock will increase if the surplus productivity is larger than the catch amount, and vice versa. If the catch amount is a lower horizontal line in Fig. 7.1, the black circle at the intersection between the horizontal line and the parabola is in equilibrium. If the stock abundance is less than the white circle, the surplus production is less than the catch and the stock is reduced until it is depleted.

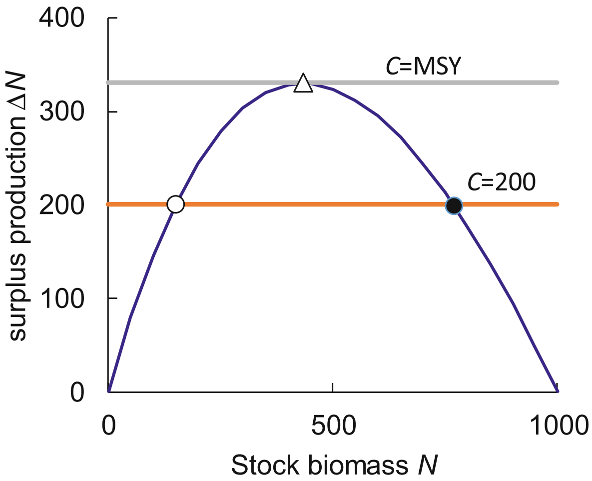

Fig. 7.1 The Relationship between bank interest rates and surplus production ($\Delta N = N_{t+1} - N_t$) of bioresources. The vertical component of the vertex (△) of the parabola gives the MSY. We assume that the expression $N_{t+1} = N_t \exp\,[r\,(1 - N_t/K)] - C$, where $r = 1$, $K = 1000$; r is the maximum growth rate (or the interest rate in the context of bank saving), and K is the carrying capacity when no fish are caught, which is unlimited in bank saving

To maximize catch at equilibrium, it is possible to catch corresponding to the upper horizontal line in Fig. 7.1. This is called the maximum sustainable yield (MSY). However, if the amount of resources falls slightly below the equilibrium point, it will decrease until it is depleted. MSY is vulnerable to disturbance and uncertainty.

For the second difference from bank saving, the maximum growth rate in a fisheries resource is usually uncertain. If you do not know the surplus curve in Fig. 7.1, you cannot get exact MSY. Third, the biomass corresponding to the bank principal is not well known. If these are uncertain, the MSY will not work. Fourth, the growth rate of biological resources (corresponding to the interest rate) fluctuates greatly from year to year. Although we know now market-linked deposits, the fluctuations in the interest rates of biological resources are so extreme that they can diminish even without any catch. This is more like investing in stocks than bank savings. Thus, for an uncertain and changing ecosystem, the classical MSY theory, as shown in Fig. 7.1, was a desk theory.

7.2 Stock–Recruitment Relationship

As we wrote at the beginning, since the entry into force of the UN Convention on the Law of the Sea in 1996, Member States are able to use their marine resources exclusively in their exclusive economic zones (generally 200 nautical miles offshore). They were also responsible for their use and decided on the total allowable catch (TAC). In Japan, fisheries scientists assess the allowable biological catch (ABC) for seven fisheries resources, before the government determines the TAC every year in consideration of social factors.

As a modification of the MSY theory, a management rule (ABC decision rule) that changes the catch according to changes in the stock amount has been developed as shown in Fig. 7.3. Instead of the total stock abundance, the spawning stock biomass (SSB) that does not include immature fish is usually used as an indicator in ABC decision rules.

As mentioned earlier, if there is no density effect, MSY cannot be defined in principle. Instead of the surplus production relationship in Fig. 7.1, the relationship between SSB (denoted by B_t) and recruitment amount (denoted by $N_{0,\ t}$) as shown in Fig. 7.2 is often used.

$$\widehat{N}_{0,t} = RB_t e^{-kB_t} \tag{7.1}$$

$$\widehat{N}_{0,t} = \frac{RB_t}{1 + aB_t}$$

$$\widehat{N}_{0,t} = \mathrm{Min}[N_{0\,max}, RB_t]$$

where $\widehat{N}_{0,t}$ is the theoretical value of recruitment; R means the number of recruit per unit spawning stock biomass without density effect, k and a represent the strength of density effect in Ricker model and Beverton-Holt model, respectively. The recruitment proportionally increases with SSB until $B = N_{0max}/R$ and thereafter the recruitment is constant irrespective of B if $B > N_{0max}/R$ in the hockey-stick model. We call the ratio $N_{0,\ t}/B_t$ the "recruitment per spawning stock biomass" (RPS).

We chose these parameter values to minimize the sum of squares of the difference between the log of the estimated recruitment $\widetilde{N}_{0,t}$ and the log of its theoretical value $\widehat{N}_{0,t}$, i.e., $\sum_{t=1952}^{2014}\left[\log\left(\widehat{N}_{0,t}/\widetilde{N}_{0,t}\right)\right]^2$. This is called the least squares method, which is a kind of maximum likelihood method and is used when the error has a normal distribution with a constant variance. In conclusion, we obtained $(R, k) = (1, 2.1 \times 10^{-5})$ for Ricker model, $(R, a) = (2.15, 0.00015)$ for Beverton-Holt model, and Nakatsuka et al. (2017) estimated that $(R, N_{0max}) = (0.56, 30,000)$ for hockey-stick model.

Anyway, as is seen from Fig. 7.2, the annual recruitment fluctuates greatly from the theoretical value. There are two possible reasons. One is due to the estimation error of SSB and the number of recruitment, and the other is that the true RPS fluctuates due to environmental fluctuation. The latter is called process error. In Chap. 10, the process error is given by the logarithm of the ratio between the theoretical value and the true value: $\xi_t = \log\left(\widehat{N}_{0,t}/N_{0,t}\right)$; and the measurement error is given by $\zeta_t = \log\left(\widetilde{N}_{0,t}/N_{0,t}\right)$. The true recruitment $N_{0,\ t}$ is unknowable, but if the regression model is correct, the sum of the variances of the two errors is obtained. If there is no correlation between the measurement and process errors, the

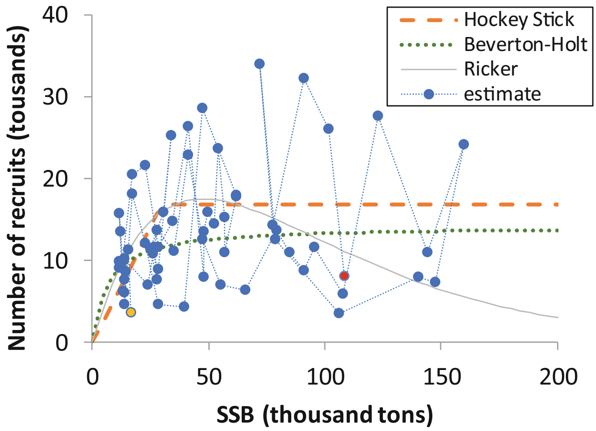

Fig. 7.2 The relationship between the estimated spawning stock biomass and the number of recruits of Pacific bluefin tuna from 1952 (red circle) to 2014 (yellow circle) (International Scientific Committee on Tuna and Tuna-like Species in the North Pacific Ocean 2016, see also Nakatsuka et al. 2017). Regression models (Ricker, Beverton-Holt, and hockey-stick) are shown by thin, dotted, and broken lines

variance of $\log\left(\widetilde{N}_{0,t}/\widehat{N}_{0,t}\right)$ is $\sigma_r^2 + \sigma_N^2$, where σ_r and σ_N represent the standard deviations of ξ_t and ζ_t, respectively. If the variance of estimation error σ_N^2 is estimated by another method, the variance of process error σ_r^2 can be estimated. However, we do not know which of the above three or other models is correct. There is also an estimation error in the parameter values. When considering resource management strategies, the results will change depending on which model is selected. Therefore, we will seek a more robust management policy by considering the uncertainty of the model as well as the measurement error and the process error. This kind of examination is conducted in Management Strategy Evaluation (MSE) in fisheries (Punt et al. 2014).

However, at least in Japan, it seems that MSE, which has been established in fisheries resource management, or alternative risk management evaluation, is not considered for new coronavirus countermeasures and wildlife management, except deer management in Hokkaido (see Chap. 10). Bayesian estimation, which is introduced in Chap. 10, is used to estimate the current situation in both fields, but future forecasts do not fully consider process errors.

7.3 Total Allowable Catch System

Fisheries scientists evaluate the stock biomass every year, and keep the catch rate constant (F_{limit}) if the stock is above a certain threshold (B_{limit}). If it falls below B_{limit}, the fishing coefficient (approximately the ratio of catch to stock) is reduced in proportion to the biomass.

The value of F_{limit} is the fishing coefficient that fisheries scientists theoretically consider to be able to maintain the stock abundance above the B_{limit}. However, for overestimation due to uncertainty, a catch rate slightly lower than F_{limit} is recommended as $F_{ltarget}$. It is usually recommended to reduce by 20% of F_{limit}. As mentioned in the previous section, the risk assessment may allow for a safety factor in this way. This is based on precautionary measures. There are several ways to determine F_{limit}. We may obtain F_{limit} that achieves MSY if we know the reproductive relationship as shown in Fig. 7.1 (F_{MSY}); we may obtain F_{limit} using the current catch rate when the current fishery can be recognized as appropriate ($F_{current}$); we can obtain F that maximizes the yield from the current cohort without considering the next generation (F_{lmax}); we may obtain F_{limit} that are expected to maintain current stocks (F_{sus}); we may obtain F_{limit} that are required to achieve a defined resource recovery plan (F_{rec}); and we may obtain F_{limit} under which the number of eggs spawned by a certain cohort is 30% of those when there is no fishing ($F_{30\%\ SPR}$).

Here, we explain %SPR (Percent spawning per recruitment, Mace and Sissenwine 1993). In order to ensure the sustainability of resources, for example, when the resource is desired to be restored, the rate of increase is set to 1 or more. Certainty is also high. The internal natural growth rate (logarithm of the maximum eigenvalue λ of the above Leslie matrix) cannot be estimated unless the initial survival rate P_0 is estimated.

But we can evaluate the following index used in fisheries science:

$$SPR = \int_{t_1}^{t_\infty} m(a)\exp[-M(a) - F(a)]da \tag{7.2}$$

where $M(a)$ and $F(a)$ represent natural mortality coefficient and fishing mortality coefficient (or fishing coefficient) at age a, respectively; t_1 and t_∞ mean the age at recruitment and the physiological longevity, respectively. SPR indicates the spawning per recruitment. If M and F are constant, e^{-M} and $e^{-M-\mathrm{F}}$ indicate the annual survival rate without and with fisheries, respectively. The total annual mortality rate is $(1 - e^{-M-\mathrm{F}})$, of which the ratio of fishing coefficient to natural mortality coefficient is F:M, so the catch rate D per year is

$$D = \frac{F}{F+M}\left(1 - e^{-F-M}\right) \tag{7.3}$$

In Eq. (7.2), the number of eggs spawned by age a, $m(a)$, is used. The egg production SPR decreases as the fishing coefficient increases. The SPR without fishing is expressed as $\mathrm{SPR}_{F=0}$, and the percentage of the SPR with fishery to the SPR without fishery is expressed as %SPR.

$$\%SPR = \frac{\int_{t_1}^{t_\infty} m(a)e^{-M(a)-F(a)}da}{\int_{t_1}^{t_\infty} m(a)e^{-M(a)}da} \times 100\% \tag{7.4}$$

This quantity is an indicator of overfishing. It is usually said that %SPR should be 30% or more. Fisheries deprive wildlife breeding opportunities by 70%. If the %SPR is around 30%, we expect that sustainable resource use would be possible. The

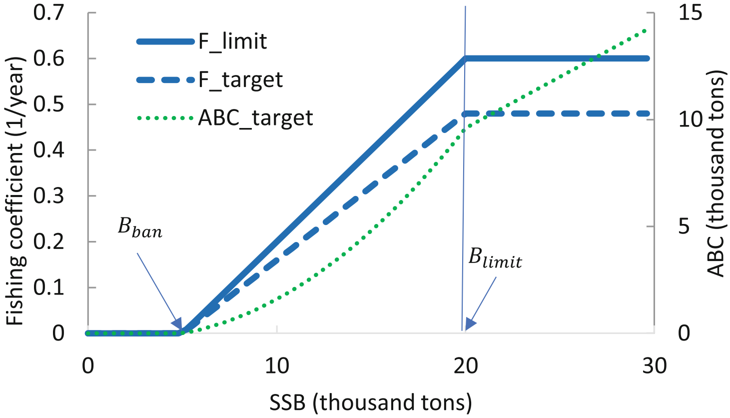

Fig. 7.3 Schematic diagram of the rules for determining allowable biological catch (ABC) by the Fisheries Research Agency (2020). It is designed to reduce the catch rate when the stock decreases, and to incorporate a safety factor by precautionary measures

number of spawns is often unknown. If we assume $m(a)$ of matured fish is proportional to the body weight, $m(a)$ would be substituted by the product of the body weight $w(a)$ and the maturity rate.

However, the allowable value of %SPR depends on the fish species. Thirty percent is a feasible target in times when there were many fisheries that were below that level. %SPR should be kept high for large predators such as tuna, and relatively low for small prey such as sardines. In addition, because the natural abundance of small prey populations varies significantly, it should be considered separately in high-level periods and low-level periods, or in the dominant cohort and other cohorts. Decision of a uniform %SPR value is not always effective for fisheries management.

It is questionable whether to take precautionary measures when resources are not overfished. As introduced in Chap. 1, the Rio Declaration has the proviso "for serious or irreversible damage." Resource loss due to overfishing is likely reversible. In fact, F_{target} in Fig. 7.3 has rarely been applied in Japan, except for Pacific saury, which is actually making production adjustments. What is needed is a precautionary measure to avoid the serious impact of resource depletion, as shown in Fig. 7.3, that fishing is prohibited when the number of fish falls below a certain standard (B_{ban}). Until 2002, there was no B_{ban} in Japanese management rules, and fisheries could continue even if resources were collapsed. It is not possible to adequately prevent the depletion of resources without B_{ban}.

There are critics of taking precautionary measures even when resources are sufficiently abundant. As introduced in Chap. 1, the Rio Declaration has a proviso "on serious or irreversible damage." Resource reduction due to overfishing may not be irreversible. In the early stage of TAC system in Japan, F_{target} had not been applied, except for Pacific saury, which is actually making production adjustments. What is needed is a precautionary measure to avoid the serious impact of resource depletion, where fishing is reduced to below a certain threshold (B_{ban} in Fig. 7.3). Until 2002, there was no B_{ban} in the ABC decision rule, so that fisheries could continue no matter how low the stocks were. This is not enough to guarantee sustainable fisheries.

7.4 Adaptive Risk Management in Fisheries Resources

The MSY theory that assumes steady-state and complete information is easy to understand as a philosophy but is hardly useful for real environmental issues. Therefore, the concept of adaptive management that presupposes uncertainty and unsteadiness has emerged. Adaptive management reduces the risk of management failure by implementing a management plan based on unverified assumptions, continuously monitoring the validity of the assumptions, and changing measures in response to status changes. The process of verifying the assumptions and, if necessary, modifying them is called adaptive learning. Changing the policy according to the state change is called feedback control. Adaptive management consists of adaptive learning and feedback control.

Adaptive management shares a common characteristic with the precautionary principle in that management is performed before the assumptions are scientifically confirmed. Compared to the precautionary principle, the post-verification process is explicitly emphasized in adaptive management. Also, while the precautionary principle is often interpreted as regulating unless safety can be demonstrated, it is important that adaptive management be enforced even if it has not been confirmed, and that verification work be performed after the enforcement.

Important points in adaptive management are (1) to clarify the assumptions used, (2) to determine in advance algorithms of how to change policies according to state changes, (3) to establish evaluation criteria, and (4) to incorporate uncertainties, (5) to prepare to handle a variety of situations, (6) to build trust among stakeholders, and (7) to realize a possibility that current decision is wrong (Matsuda and Nishikawa 2008). Everything does not go as planned, but it is important to thoroughly consider the feedback control method in advance and to take measures against various situations without expecting a single future.

7.5 Risk Management of the Allowable Biological Catch Decision Rule

It is desirable to have a management rule that takes into account various uncertainties, prevents severe resource depletion, avoids ban on fishing as much as possible, and obtains a high average catch. Considering the annual variation of the recruitment rate (r in the resource dynamics equation as mentioned above) and the

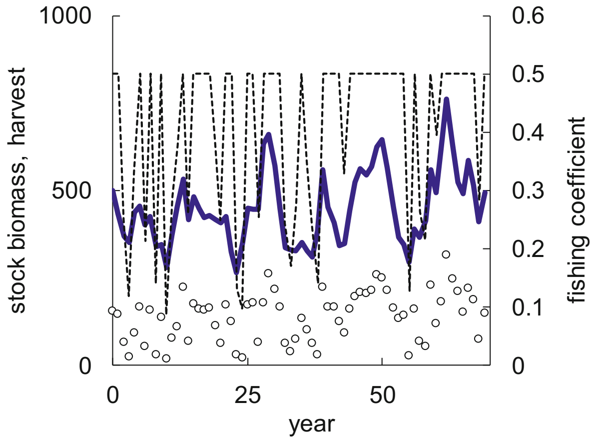

Fig. 7.4 Fisheries management model by numerical experiments using stochastic simulations. The bold line indicates the change in the stock abundance. The dotted line indicates catch amount, circles indicate catch

measurement error of the stock biomass assessment, research has been conducted on how to determine a desirable ABC decision rule (Katsukawa 2004). Here, we introduce an example using a simple mathematical formula. Depending on B_{ban}, B_{limit}, and F_{target} in Fig. 7.4, the risk of falling below a certain resource threshold (for example, the abovementioned 5000 tons) and the risk of implementing fishing bans are limited to a certain limit, for example, the risk of management failure and the performance of average catch and minimum stock (Fig. 7.4). From the author's website (https://ecorisk.web.fc2.com/springer2021.html), you can get a Microsoft Excel file that relives the calculations in Fig. 7.4.

Suppose that the resource amount N_t fluctuates yearly by the following equation. This corresponds to a fishing season immediately before the spawning season.

$$N_{t+1} = (N_t - C_t)\exp[r_t - \alpha(N_t - C_t)] \tag{7.5}$$

Since the resource estimation involves a measurement error, we assume that the estimated stock abundance, denoted by $\tilde{N}_t$, is described:

$$\tilde{N}_t = N_t(1 + \sigma_e \xi_t) \tag{7.6}$$

where ξ_t is a standard normal random variable; σ_e means the SD of measurement error, assuming 30% here. According to this estimated value, we decide the fishing coefficient, denoted by F_t, as shown in the graph of Fig. 7.3, and the allowable catch $\hat{C}_t$ and we assume the actual catch C_t as follows.

$$\widehat{C}_t = \tilde{N}_t F_t[1 - \exp(-F_t)] \tag{7.7}$$

$$C_t = \mathrm{Min}[\widehat{C}_t, N_t F_t(1 - \exp[-F_t(1 + \sigma_c \zeta_t)])], \tag{7.8}$$

where ζ_t is a standard normal random variable; and σ_c means the implementation error. The allowable catch is calculated from the estimated stock and the catch coefficient, and the catch does not exceed it. However, when the actual stock is low, the catch often falls below the allowable catch $\hat{C}_t$.

In Fig. 7.4, we show a stock fluctuation obtained by a simulation experiment where we assumed that $r = 0.5$, $K = 1000$, $\sigma_r = 1$, $\rho = 0.7$, $\sigma_e = 0.3$, $\sigma_c = 0.1$, $F_{target} = 0.24$, $B_{ban} = 200$, $B_{limit} = 600$.

To reduce fishing bans, B_{ban} in Fig. 7.3 should be as low as possible, or zero. However, the risk of abundance below the threshold will increase, and both the average catch and minimum abundance decrease when B_{ban} is lower. Even when B_{ban} is set high, the fishing rate is suppressed early, it is possible to halt the decline in stocks, and as a result, the minimum stock and the average catch are kept high if B_{limit} is set high.

Reducing the catch rate when the stock is low will reduce temporal catches, while it will prevent unnecessary stock loss and maintain high catch amounts in the long run. However, it is hard to reach agreement with fishers about this. Some of the causes are listed below.

As mentioned above, sardines undergo natural fluctuations. The catch amount may increase even though the fishing coefficient is too much, while it may decrease even if the fishing is prohibited. Therefore, the effect of management is often unclear. However, those who make financial investments must understand gambling. Fishers who are good at gambling must understand the stochastic management in fisheries. In some sections, fisheries scientists misunderstand the recommendation based on classical MSY theory. Even if the fishery stock fluctuates, fishery management is necessary. In fact, fishers are under the asylum of the fisheries administration, and compensation is often obtained if the resource is reduced. In particular, if the state proposes to reduce the number of fishing vessels, compensation will be obtained. In this case, fishers will catch as much as they want and rely on the government if the stock and catch decrease.

Thus, in fisheries management, risk management is definitely important, which takes into account uncertainty and variability. The fisheries management makes future predictions using stochastic models and calculates the probability of stock recovery or stock collapse and reflects these results in policies. We need to set resource recovery goals, but it should be set by agreement of stakeholders, rather than by scientists. It is important that the progress of risk management methods and the creation of a framework for consensus building in parallel (Figs. 7.5 and 7.6).

7.6 Regime Shift and Process Errors

However, we should take care when detecting the density effect in biomass between the present and next year. Suppose that a time series $\{N_t\}$ is generated by a Ricker model: $N_{t+1} = N_t e^{r+\xi_t-kN_t}$. If we draw a scatter plot between the log of biomass $x_t = \log N_t$ and the log of increasing rate $\rho_t = x_{t+1} - x_t$, we usually find a

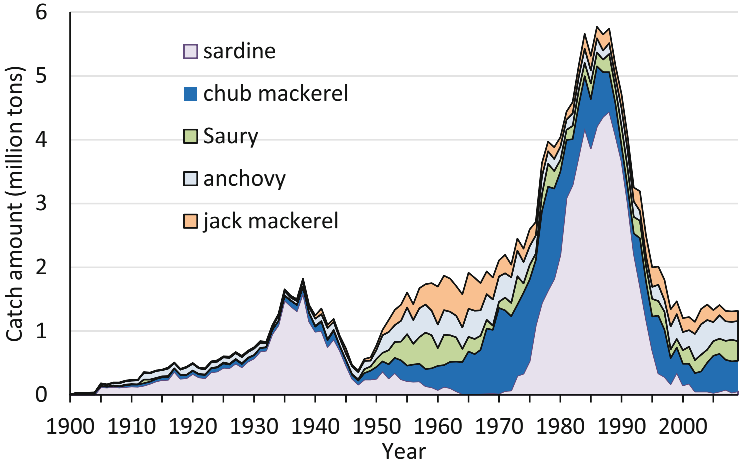

Fig. 7.5 Catch statistics of major small pelagic species in Japan (Ministry of Agriculture, Forestry and Fisheries, Japan, updated from Matsuda and Katsukawa 2002)

significantly negative correlation between x_t and ρ_t, even when $r = k = 0$ and $\sigma_r > 0$, or density effect does not exist but process error exists. We call this "fake density dependence" (Dennis and Taper 1994; see the Excel file for detail).

We can intuitively explain as follows. In the past time series, $x_{t+1} < x_t$ in the year following the year with the largest x_t, so ρ_t is negative, and ρ_t when x_t is the smallest must be positive. Therefore, in the past time series, a year in which x_t is large tends to decrease in the next year, and vice versa. So even if there is no density effect, a negative correlation is likely to occur between x_t and ρ_t. The method of statistically detecting the density effect from an observed time series is based on the correlation coefficient between x_t and ρ_t of many artificial time series generated by randomly shuffling the order of ρ_t obtained from the observed time series. If the correlation coefficient of the given time series is significantly smaller (larger in the absolute value) than that of the artificially generated time series, the observed data has a density effect, because the randomly generated data must have no density effect (Dennis and Taper 1994).

Similarly, when detecting the density effect from the time series, it is necessary to be careful whether the errors can be regarded as independent or not. Especially for fisheries resources, it is said that the annual process error is not independent, and there is a regime shift in which the sea surface temperature on spawning grounds between a high water temperature regime and a low water temperature regime in units of about one decade. Ignoring the regime shift in such cases can lead to strange interpretations of stock assessment.

Figure 7.7 shows a scatter plot with the horizontal axis representing the ratio of the estimated SSB B_t in each year to the biomass B_{msy}, which is the MSY level; and the vertical axis representing the ratio of the estimated fishing coefficient F_t and the F_{msy} that is expected to achieve the MSY level. This plot is called "Kobe plot"

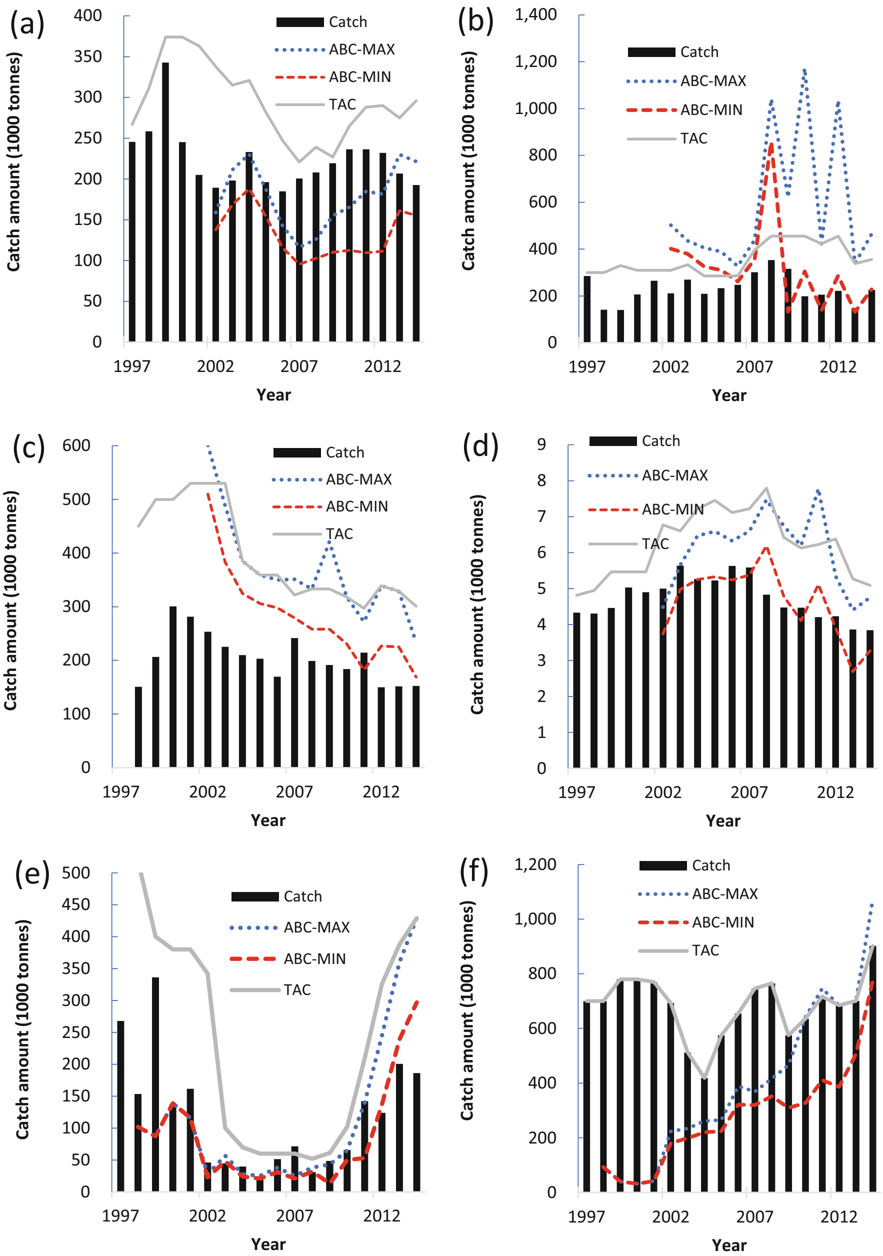

Fig. 7.6 Annual changes in the total allowable catch (TAC), maximum and minimum allowable biological catch (ABC), and actual catch amount of major six species, (**a**) walleye pollock, (**b**) Pacific saury, (**c**) squid, (**d**) snow crab, (**e**) sardine, and (**f**) chub mackerel. TAC of sardine in 1997 and 1998 are 720 and 520 thousand tons. ABC of some species during 1997–2001 are not disclosed

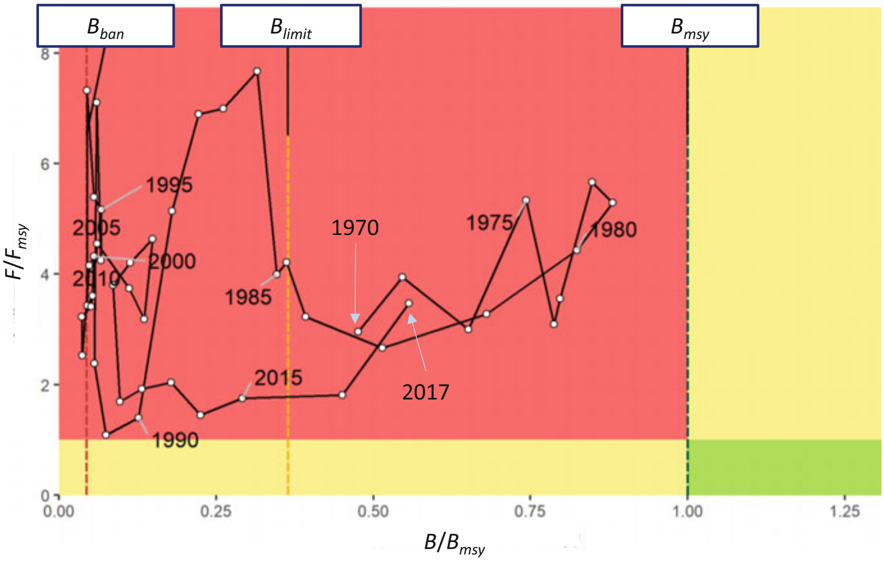

Fig. 7.7 Scatter plot of Pacific stock of chub mackerel (National Research Institute of Fisheries Sciences 2019)

because usage of this idea is agreed in the Joint Meeting of Tuna RFMOs (regional fisheries management organization) held in Kobe, Japan, 2007 (Maunder and Aires-da-Silva 2011). The resource status can be evaluated by (B_t, F_t) in each year. In this figure, if $B_t/B_{msy} < 1$, it indicates that this stock has been overfished; and if $F_t/F_{msy} > 1$, it indicates that the current catch is excessive (overfishing). In other words, the red region in the upper left indicates overfished in the past and overfishing in the present, while the green region in the lower right suggests that more fish can be caught sustainably. The Kobe plot is frequently used in other articles (e.g., Worm et al. 2009; Ichinokawa et al. 2017).

However, the evaluation method of B_{msy} and F_{msy} is not always appropriate. In this figure, it was assessed as overfished and overfishing even in 1972, when the stock was the highest, and it was estimated that overfishing continued despite the fact that the stock steadily increased from 1999 to 2016. As will be shown later, the chub mackerel stock was at a high level in the 1970s, whereas it was collapsed around the 1990s due to both overfishing and the lower regime of RPS. Assuming an independent process error in RPS throughout the period, it cannot be explained that the stock recovered smoothly.

In the case of basic research, an unrealistic analysis that is mathematically consistent will do little harm. However, fisheries management is an economic and social painful practice for fishers and other stakeholders. The theory is not always right. It will be very confusing if we do not always match reality and examine the validity of the assumptions used in that theory.

7.7 Overfishing of Sardine and Chub Mackerel in Japan

Sardine was at a high level during the 1930s and 1980s and at a low level during the 1960s in Japanese water (Fig. 7.5). During the rapid decline around 1990, there was almost no new recruitment from 1989 to 1992, and the age structure of sardine was biased in older. Therefore, the decline is not due to overfishing but to natural fluctuations. If reduced due to overfishing, the age structure will be young. In 1988, sardines alone caught 4.5 million tons per year, and Japan's total catch was 12 million tons, boasting the highest catch in the world. In 2003, the catch of sardine decreased to about 50 thousand tons. The B_{ban} of the Tsushima warm current stock (population on the Sea of Japan side) is set at 5,000 tons. This is based on the lowest levels of estimated resources in the 1960s. We aimed for management to prevent from further decline below the historically lowest level.

The ABC in Japan was not disclosed in 1997 but has been published since 1998. Looking at the sardine in Fig. 7.6, it can be seen that the TAC was always significantly higher than ABC, and the actual catch was also higher than ABC. It is inevitable that resources continued to decrease.

Japanese Fisheries Research Agency (FRA) defined 5000t as B_{ban} of the Sea of Japan stock of sardine. In order to maintain the stock more than B_{ban}, FRA and the Fisheries Conservation Division of Fisheries Agency had previously set a low ABC based on Fig. 7.3. However, the TAC was set several folds higher than that, and the actual catch was higher than ABC. As a result, the stock continued to decline, and finally a report was given in 2004 that fell below B_{ban} and had an ABC of 0. A strong objection was raised at this time, especially from sardine fishers on Pacific coast. The ability to comply with ABC in resource management is an important point for Japan's future fisheries management.

We have warned that overfishing of mackerel reduced both resources and catches (Matsuda et al. 1992). Kawai et al. (2002) showed that overfishing in the 1990s hindered resource recovery (Fig. 7.8). At that time, the probability of stock recovery when the overfishing was continued in the future was calculated. Of the 1000 trials, none of the trials had recovered to 1 million tons by the next 20 years. On the other hand, when fishery focused on adult fish was conducted as was done before 1990, the probability of stock recovery was about 50% (Fig. 7.8).

After we showed these results to fishers, the fishers also began to recognize the necessity for fisheries management of chub mackerel. The FRA's stock assessment report also states that overfishing of the preeminent year class in 1992 and 1996 prevented the stock recovery.

As can be seen from Table 7.1, the FRA calculated ABC based on several management scenarios and did not make a single forecast of future inventory for each scenario. The FRA reported the risk that numerical targets were not met and indicated the probability of management failure. Which policy to use is up to the social agreement. In this sense, the concept of risk management has already been established in the Japanese TAC system.

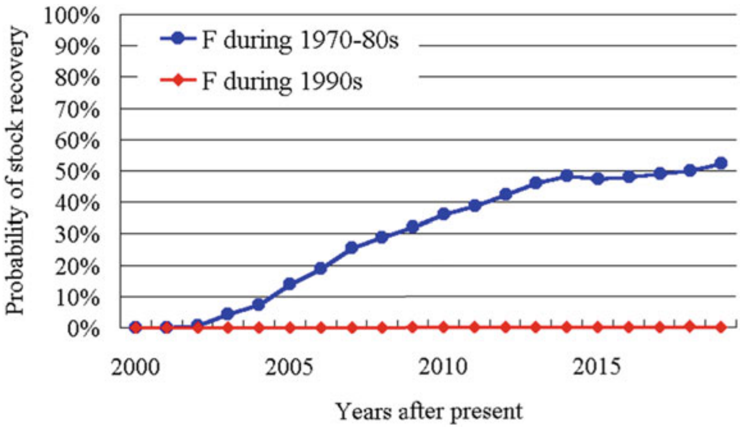

Fig. 7.8 The probability of stock recovery when the mackerel population was caught mainly in adult fish as was done in the 1980s, and when it was caught mainly in immature fish as was done in the 1990s (modified from Kawai et al. 2002)

Table 7.1 Fisheries management scenario for Pacific stock of chub mackerel (the digest version of the 2006 Fisheries Research Agency http://abchan.job.affrc.go.jp/digests18/html/1809.html)

Scenario	Management policy	ABC in 2007	F	Harvest ratio	Evaluations
ABC limit (F_{rec})	To recover stock by reducing fishing ecoefficient	54,000 tons	0.31	21%	A:39% B:100% C:187,000t
ABC target ($0.8F_{rec}$)	Precautionary measure of the above scenario	46,000 tons	0.25	18%	A:52% B:100% C:191,000t
Current catch amount (F_{sus})	Spawning stock biomass has been maintained at a constant level of 80,000 tons since 2008.	93,000 tons	0.7	36%	A:1% B:53% C:119,000t
Current fishing coefficient ($F_{current}$)	Current fishing coefficient is maintained during 2003–2005	85,000 tons	0.61	33%	A:2% B:55% C:126,000t

1. The fishing coefficient F is a simple average for all ages;
2. Harvest ratio = ABC/B, where B is the average of estimated stock biomass as of TAC calculation in July and the previous year
3. $F_{current}$ is the average F during 2003–2005
4. A to C in the evaluation column show the simulation results considering recruitment and uncertainty of stock assessment. A: Probability that $B_{2014} > B_{limit}$; B: The probability that $B_t > B_{min} = B_{2002} = 37{,}000$ tons; C: the average catch between 2007 and 2014.

7.8 International Fisheries Management of Pacific Saury

Pacific saury is a target species that defines TAC under the law in Japan. As is seen in Fig. 7.6, unlike other fish species, the TAC is lower than the ABC and the actual catch is lower than TAC. The demand for saury is limited and large fish are mainly used as grilled fish. Fishers adjusted the catch effectively, as the price collapsed if the catch was too high. However, saury stock was almost at its highest in the 1990s and 2000s. According to the United Nations Convention Law of the Sea (UNCLOS), if the catch is less than the TAC, foreign fishing vessels cannot be excluded. Exclusive Economic Zones (EEZ) may have quotas assigned to other countries if coastal countries have the right to prioritize sustainable use of resources and catches below the TAC.

The Pacific saury is a highly migratory fish and is distributed in the Kuroshio Extension basin beyond the exclusive economic zone, so foreign fishing vessels such as Taiwanese fishing vessels have begun to catch large quantities outside the EEZ. In Japan's coastal offshore fisheries freeze fish after landing, whereas foreign fishing vessels are new and large, and freeze fish on board.

Since the introduction of the TAC system in Japan, saury fishing vessels have been equipped with separation equipment that sorts fish according to size. Small fish are sold at low prices because they are used as feed for farmed fish. For these reasons, the TAC is designed to prevent oversupply, and small fish are used to catch large fish within the TAC. There was a suspicion that a small fish was dumped in the sea. Similar problems can arise with the TAC system, as there is no regulation on length composition and only the total amount is regulated without counting discarded fish. Due to the small scale fisheries, it is not practical to put watchmen on fishing boats. It would have been possible to ban devices that could sort small fish.

This has caused a rebound from fish processors who used small fish. Later, edible large fish with limited demand fell into oversupply, and the fish prices of large fish collapsed. Therefore, National Saury Fishery Association stopped their member fishers from installing separators (Oyamada et al. 2009). The Pacific saury stock was at a high level in the 1990s, and it cannot be said that the dumping of small fish had a direct impact on stocks.

As mentioned above, there are various problems with the system of the TAC system. In some cases, the incentive to avoid overfishing may not work properly, and may even increase discarded fish. An international management framework is indispensable for fish stocks to distribute beyond the EEZ. First, it is necessary to make a consensus on an international agreement that eliminates the adverse effects of large-scale dumping of small fish. We hope fishers lands all of the small fish even if they can sort small fish.

Japanese fisheries are free trade in principle, and it is difficult to distinguish between overfished and managed marine products if they are sold in the market. Setting a certain environmental standard for the production of agricultural, forestry, and fishery products and importing only those that meet the standard can protect the

Japanese fishery to some extent. The non-tariff wall could at least limit imports of aquaculture products made by overfishing or destroying coastal ecosystems. Of course, even if environmental standards are set, it is not possible to restrict the import of environmentally friendly fisheries products from overseas. However, it would be great if Japan, the world's largest seafood importer, could contribute to protecting the world's seas by setting environmental standards.

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Chapter 8
Why Is the Tuna Critically Endangered and Still Sold in the Market?

Hiroyuki Matsuda and Mitsuyo Mori

Abstract In 1996, the International Union for Conservation of Nature (IUCN) designated southern bluefin tuna as a critically endangered species, which is the highest endangered category of threatened species. Threatened species in IUCN's Red List is determined based on five quantitative criteria, which are based on precautionary principle for organisms whose populations are not well known. When these criteria are applied directly to a commercially exploited species, it meets the criteria even if it is apparently secure. This section introduces the concept of judgment criteria for endangered species and its concerns. Threatened species is listed if the reduction rate of population size is sufficiently large, irrespective of whatever large population size. This is because for most unexploited threatened species the population size is completely uncertain. Therefore, many marine fish species, including tuna, which are exploited by commercial fishing are listed as threatened species. For many fish species that are commercially exploited, the absolute number of mature individuals and the variance in its yearly reduction rare are often known. Extinction risk depends on the absolute population size, population reduction rate, and its temporal variation. Red List lists many species that are apparently secure, based on the precautionary principle. We propose a method for estimating the extinction risk of tuna based on the variance of the reduction rate. We investigated the sensitivity in the uncertain parameters involved in the models and concluded that tuna is unlikely to be listed as critically endangered but that southern bluefin tuna may be listed as vulnerable. We also propose the probability that the southern bluefin tuna is recovered by the year 2020, which is not optimistic. Finally, we obtain the optimal age-specific fishing policy by using the Lagrange method.

Keywords Red List · Extinction risk · Precautionary principle · Population viability analysis · Inverse baby-boom effect

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8.1 Red List Categories and Criteria

In 1996, at the IUCN World Conservation Congress, southern bluefin tuna (*Thunnus maccoyii*, SBT) and the western population of Atlantic bluefin tuna (*Thunnus thynnus*, ABT) were listed as the threatened species. Threatened species in the IUCN Red List are divided into three categories according to their extinction risk, referred to as Critically Endangered (CR), Endangered (EN), and Vulnerable (VU) (Fig. 8.1). The Red List means the list of threatened species. The criteria to determine threatened species is called Red List criteria.

The Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES), which restricts the international commerce of endangered organisms, also lists each organism according to the extent of its threat of extinction. CITES lists species in its Appendix I, II, or III, in terms of the degree of threat. International trade of a species is prohibited if it is listed in CITES Appendix I. A species listed in CITES Appendix II requires a permission document by the export country for international trade. If a nation seeks help from another country to protect a species, the nation includes this species in Appendix III.

IUCN revised the Red List criteria in 1994 and adopted quantitative criteria. IUCN Red List criteria are applied to all taxa of plants and animals from land to marine. However, some criticism arose in the process. When IUCN listed marine fisheries resources in 1996, some species that were clearly not at risk of extinction were listed, and criticism of grading and criteria was increased. The SBT mentioned here is one of them. Therefore, IUCN decided to review the criteria at the same time

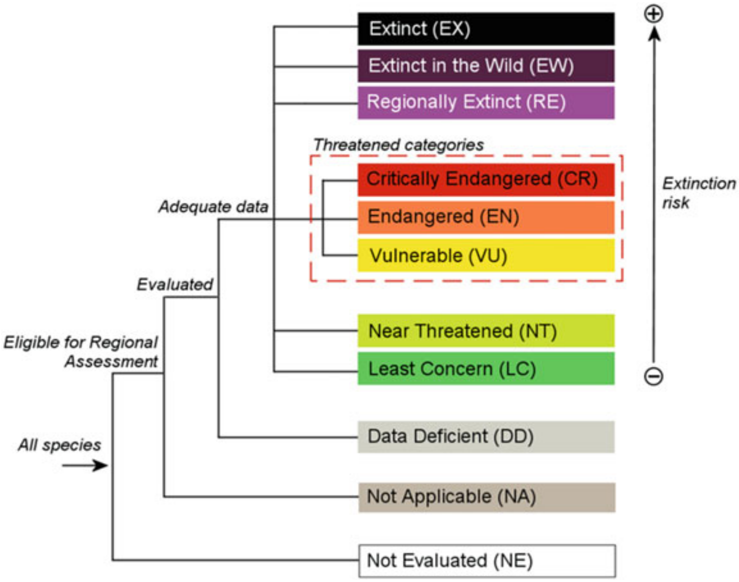

Fig. 8.1 The IUCN Red List categories at the regional scale (IUCN 2012)

Table 8.1 Summary of IUCN Red List criteria (IUCN 2001)

Criterion	CR	EN	VU
A: Population decline rate is	>80%/10 years or three generations	>50%/10 years or three generations	>30%/10 years or three generations
A2: (under managed)	>90%/10 years or three gen.	>70%/10 years or three gen.	>50%/10 years or three gen.
B1: Area of occupancy is	<10 km^2	<500 km^2	<2000 km^2
B2: Extent of occurrence is	<100 km^2	<5000 km^2	<20000 km^2
C1: Population is declining and	<250 (25%/ 3 years or one gen.)	<2500 (20%/ 5 years or two gen.)	<10,000 (10%/ 10 years or three gen.)
D1: Population size is	<50	<250	<1000
E: Extinction risk is	>50% in 10 years or three gen.(cap 100 years)	>20% in 20 years or five gen. (cap 100 years)	>10% in 100 years

See IUCN (2001) for detail

as approving the Red List of Marine Animals at the IUCN World Conservation Congress in 1996. Seven workshops were held from 1997 to 1999, drafting revised standards, which were approved by the IUCN World Conservation Congress in 2000. Here, to clarify the problems and accountability of the IUCN Red List (see Chap. 9), we introduce the revised criteria in 2000 and compare it with the criteria in 1994 to understand the history of the revision process.

Figure 8.1 shows the structure of categories of species with respect to the degree of threat. There are many organisms in the world whose species have not been identified. If some organisms have been identified but no one has evaluated it is the degree of threat, they fall into “Non-evaluated” (NE). Some organisms do not have the appropriate information to decide Data Deficient (DD). Some are considered extinct (EX) or extinct in the wild (EW), others are judged to be least concerned (LC).

Extinction literally means that the species has disappeared from the earth. Extinction in the wild means that anything other than those in human captivity has gone extinct. This will be judged carefully and will be placed on the list one generation time after no individual has been found in any place that has ever lived. According to the definition of IUCN, “Endangered” refers to one category of threatened species, whereas “endangered” in CITES and US Endangered Species Act means a threatened species in IUCN Red List categories.

The categories from CR to EN and VU were defined as the quantitative criteria shown in Table 8.1, accompanied by verbal explanation. In the IUCN Red List ver. 3.1 (IUCN 2001), the definition of CR states that “there is a scientific evidence that any one of the criteria A-E is met and therefore faces an extremely high risk of wild extinction.” Similarly, EN is defined as having “very high risk” and VU is defined as having “high risk.”

The 1994 standard is slightly different. For example, the definition of CR is "a taxon that is considered to be facing an extremely high risk of wild extinction in the immediate future, as long as it meets the following criteria A-E." Similarly, the definition of EN has "very high risk" in "near future", and VU means "high risk" in the medium-term future. In other words, the time urgency becomes ambiguous in the revised criteria of version 3.1, and the quantitative criteria in Table 8.1 become more important.

In the criteria A, C, and E in Table 8.1 described in the next section, the stricter the stage, the shorter the period of extinction. However, the three generations are not necessarily "in the near future" because it is determined in units of the generation time of the organism. IUCN clarified that the decision was made based on the generation time of the creature rather than the urgency felt by humans.

The Red List is also made for regional scales such as national and local governments. There are some cases where the species such as the Japanese crested ibis (*Nipponia nippon*) were reintroduced in China even though they were locally extinct in Japan in 2003. If the local population is completely isolated, it can be judged by the same criteria as the global scale. IUCN publishes the guideline for application to national and regional scale (IUCN 2012). Figure 8.1 is categories for the regional scales, where the Regionally Extinct and the Not Applicable category are added to the categories for the global scales.

8.2 Is the Tuna Really Endangered?

SBT are internationally managed by CCSBT (Convention for the Conservation of Southern Bluefin Tuna), which consists of Japan, Australia, and New Zealand. The stock abundance of SBT in 1996 had fallen by about 90% in the last 29 years, which meets the criterion A in Table 8.1 and was considered as CR (Matsuda et al. 1997).

For species for which the extinction risk itself can be evaluated, a species with a risk of extinction (extinction probability) of 50% or more after 10 years or within three generations of the organism, whichever is the longer, is classified as CR according to Criterion E. However, extinction risk assessment requires detailed information such as population numbers, rates of decline, and annual fluctuations. Many wildlife lacks such information. Therefore, the information that can be used for evaluation is based on the criterion A if the reduction rate of the number of individuals is only, the criterion D if the number of individuals is, the criterion B if the distribution area or the like, and the criterion C if the number of individuals and the reduction rate are available. A species is to be listed in the Red List if any one criterion is met.

In the case of SBT, the generation time is about 15 years. According to the stock assessment of CCSBT, the number of adult fish was more than four million in the 1960s, which decreased to 0.5 million in 1996. It was estimated that it decreased by about 90%, within shorter than the three generations, it met the criterion A of 80% or more reduction rate. However, the population whose size is 400,000 will not go extinct if the stock continues to decrease by 90% in the next 30 years, it will be 20,000 after three generations of 45 years. That is, the SBT did not satisfy that the condition does not correspond to the criterion E. However, if the population decline rate at that time continues in the future, it will be estimated that the number will be 500 or less in 100 years, and it will meet the criteria E for VU.

At the IUCN Working Group meeting in Cambridge, UK in June 1999, it was addressed that species with insufficient information and for which extinction risk cannot be assessed is to be listed on other criteria based on the precautionary principle, but we argued that species with known information and a low risk of extinction should not be listed as threatened using other criteria. About one-third of the members selected by IUCN agreed with us. However, the majority opposed prioritizing criterion E. The debate stuck and a majority vote rejected our proposal. As a result, the IUCN Red List inclusion criteria were slightly revised in 2001, but SBT was still listed as CR. We still think the criteria are a misuse of the precautionary principle (Mrosovsky 1997).

How to determine these criteria will be explained in the next section. Let us consider whether SBT meets the Red List criteria or not. If the annual change in the population is estimated as shown in Fig. 1.3, the extinction will be based on the future prediction of the population trend. Matsuda et al. (1998) showed the future projection and the cumulative extinction risk, or the probability that the population size is below the threshold. "Projection" used here means a weaker concept than the future prediction, and it does not mean to guarantee that future projections will realize, but means calculation result under some assumption, including that the past decline trend will continue in the future.

Since we usually consider the autocorrelation of the change in the population in the extinction risk assessment, Matsuda et al. (1997) projected the SBT did not decrease much but may even increase in circa 2000. They assumed 500 individuals as a threshold of extinction because tuna form a school (Matsuda et al. 1997).

The estimated population size of SBT was 900,000 in 2000, and it is unlikely that they will become extinct in several decades, even under the assumption that the past decline rates will continue. The generation time of tuna is thought to be about 15–18 years, and the risk that the population will be below 500 individuals after three generations is very unlikely. However, the population will be below 500 in one century future if the past decline rate will continue. Therefore, Matsuda et al. (1997) concluded that criterion E for VU, not CR, is met for the SBT.

CITES is an international convention, and a species listed in Appendix will be accompanied by legal restrictions. Therefore, Member States have reservations that do not comply with the national restriction of a listed species. However, the Red List is judged by IUCN, a "non-governmental organization," and it is not immediately subject to regulation. So there is no reservation right of any country. Therefore, it is understandable to judge scientifically without political consideration. However, it is not scientific to post even those that are known to have a low extinction risk.

A similar example is the "90% reduction theory of tunas" (Myers and Worm 2003). They concluded that the stock of tunas decreased by 90% under the assumption that CPUE is proportional to the stock abundance. Many tuna experts disagreed

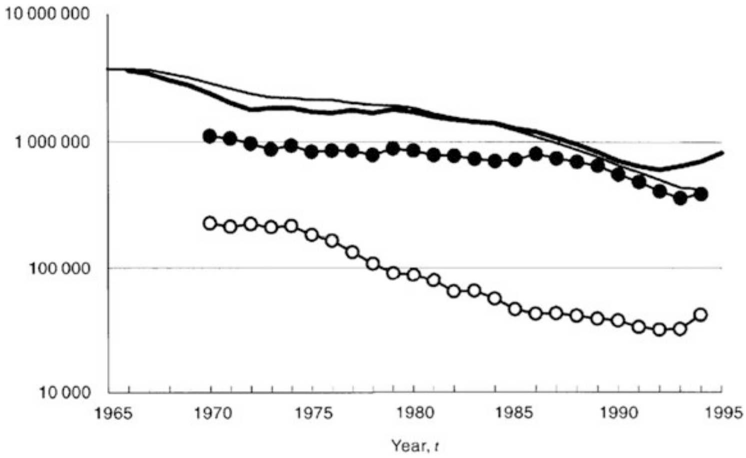

Fig. 8.2 The estimated parental stock number of the southern bluefin tuna (bold and thin lines, Ishizuka et al. 1995; Polacheck et al. 1995) and that of the western and eastern Atlantic bluefin tuna (open and solid circles, Matsuda et al. 1998)

with this published article. It was neither a tampering with the data nor a mathematical mistake, but it was a conclusion based on an improper assumption. Similarly, as an extreme claim, there is a "global fisheries resource collapse hypothesis" that the world's marine resources will be exhausted in 2048 (Worm et al. 2006). This is also an extrapolation applied by a regression equation in which the rate of decrease in the past accelerates. Although it is not mathematically incorrect, the validity of the assumption has not been verified. Even with the precautionary principle, it will be necessary to make some predictions that can withstand some verification after the fact.

It is not known which assumption is the most correct. Some looks pessimistic, others expect an optimistic future. However, the future cannot be projected without deciding an assumption. For future projection in extinction risk assessment, it is necessary to clarify what assumption is adopted and keep in mind that the assumption may not be correct. The IUCN criteria are based primarily on the extrapolation of past decline rates into the future. Figure 8.2 is also based on the assumption that the declining trend for the past 30 years will continue. In fact, if international management is successful, this premise would overestimate the risk of extinction. However, if international management fails and overfishing continues, this assumption could lead to underestimation.

The Southern Bluefin Tuna, Atlantic cod, etc., triggered the IUCN World Conservation Congress to consider revising the criteria. It was true that SBT was overfished, and those who did not eat SBT did not resist making strict judgments according to the criteria. However, it was clear that the risk of extinction was low. Following an international study group on extinction risk assessment in Tokyo in January 1999 (Matsuda 2000), IUCN organized a study group on marine life. As a result, it was recognized that it is important to collect and properly manage

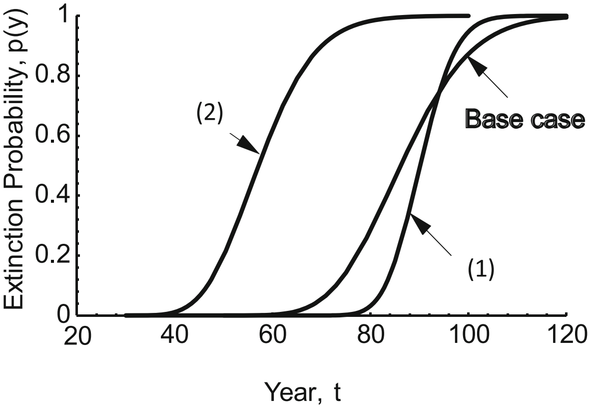

Fig. 8.3 The extinction probability of the southern bluefin tuna as a function of years (Matsuda et al. 1997). In the Base case, the critical size is assumed to be 500 and the population decline rate during the past 30 years will continue in the future. In the cases of (1) there is no autoregression in *r(t)*, (2) the critical size is assumed to be 5000

information on the target organisms and to use that information for judgment. For data-poor species, a species that satisfies any of criteria A-E is to be listed as threatened species. It leads to all environmental policies, but it is too late to take measures after full scientific certainties are obtained. Therefore, even if it is not always possible to prove that the risk of extinction is high, as in criteria A-E, it is decided to be listed (Figs. 8.3 and 8.4).

This concept is called the precautionary principle. The Precautionary Principle is stated in the Rio Declaration of the United Nations Conference on Environment and Development (Earth Summit) in Rio de Janeiro, Brazil, in 1992, and states that "(in) order to protect the environment, the precautionary approach shall be widely applied by States according to their capabilities. Where there are threats of serious or irreversible damage, lack of full scientific certainty shall not be used as a reason for postponing cost-effective measures to prevent environmental degradation."

The precautionary principle was also mentioned in a report by the Food and Agriculture Organization of the United States (FAO) (Garcia 1994). Overfishing of the fishery would not be irreversible, and that stopping the fishery may restore the stock, so fisheries management may not be necessary to incorporate the precautionary principle. However, uncertainties need to be addressed, and in the 1995 International Agreement on Stradling Stocks and Highly Migratory Fish Resources, precautionary management criteria were based on agreed scientific procedures.

The precautionary principle is now used for global environmental issues in general. Global warming and the toxicity of environmental chemicals had not yet been fully proven. However, if we wait for scientific proof and start taking measures, these global environmental problems are too late.

A similar idea is the non-regret policy. The success or failure of that policy is uncertain, but it is such a policy that there is no great regret if it fails. A typical example is the rise of women's status and education in population control measures. The aim is that the higher the education level of women, the higher the age of childbirth and the lower the birth rate. If the status of women rises and advances into society, the birth rate will decrease accordingly. However, it depends on the gender

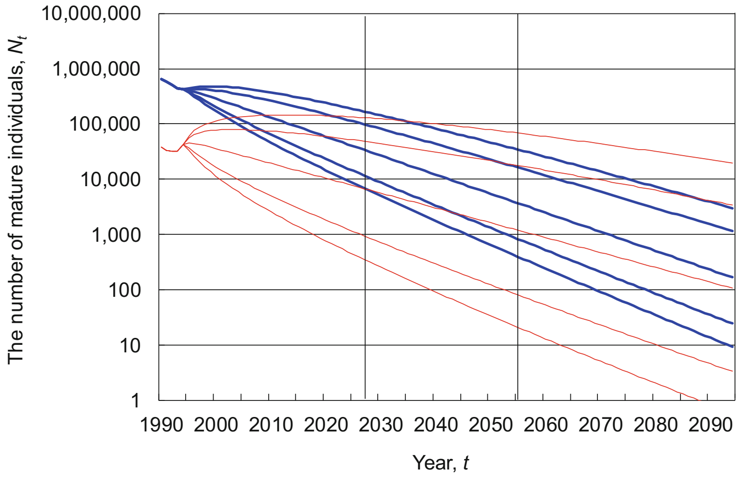

Fig. 8.4 Interval estimate of future population size, the point estimate and 95% confidence and 99% confidence. Bold lines and thin lines are respectively for the southern bluefin tuna and the western Atlantic bluefin tuna. Vertical lines in 2030, 2056, and 2096 indicate the end of the period stated in CR, EN, and VU, if $T = 12$ (Matsuda et al. 1997)

situation. As men's orientation toward childcare and the development of daycare facilities progress, women's status may be improved, and the birth rate may increase. It is not clear how effective the empowerment of women is in controlling the population. But aiming for the empowerment of women is welcome and a no regret policy. Anyway, global population control is not part of the Sustainable Development Goals.

Conversely, some policies could be regrettable. One example is the technology of fermenting grain to purify ethanol for cars. If the global food shortage is more serious than the energy crisis in the future, this technology will be wasted and regrettable.

Where endangered creatures and their habitats are used for industry, the beneficiaries have a burden of proof of sustainability. Resource users must collect and provide information. The user has the burden of proof. If lack of information is the reason for not being listed in threatened species, the user will not collect the information or provide it. Criteria should be developed that encourage and mandate information gathering. Criterion E needs more information than other criteria. I think that giving priority to criterion E is reasonable in that sense.

In addition, the purpose of the Red List is to sound an alarm to endangered creatures and encourage them to make conservation plans, and to underestimate or overestimate extinction risks depending on differences in life history. I acknowledge that there is a case to be evaluated. In addition, the purpose of the Red List is to sound an alarm to threatened species and encourage to make conservation plans. There is a case to underestimate or overestimate extinction risks depending on differences in life history.

8.3 Is the Tuna Recovering on Time?

The international management of SBT had no consensus between 1998 and 1999. Despite the opposition from Australia and New Zealand, Japan alone increased its quotas. The three countries agree that fishing grounds were well examined and began to increase. However, while Japan believes that populations outside of fishing grounds have recovered as well, the other two countries are pessimistic. There was a conflict between the Japanese side calling for a research catch outside the fishing ground to confirm this and the other two countries opposing an increase in quotas.

According to the estimates in 1997 (Fig. 8.2), although adult fish have begun to increase, immature fish have been sluggish again. Why? This is not due to management failure. In fact, the immature fish around 1995 was born around 1990. At that time, there was a period when the number of mature fish was the least. Since the catch of immature fish is refraining since 1989, the survival rate would have been better. However, the number of immature fish began to decrease again due to the small number of spawners around 1990.

Therefore, SBT stock may not recover monotonically. There is a concern that the recovery slows down again and even the stock begins to decrease. If the management is successful, it will increase in the long run, but there could be a wave of recovery.

This can be said to be the reverse of the population trend of the Japanese. The baby boomers are children whose birth rate has increased significantly after the second world war. About 30 years later, the second baby boom happened. At the second time, the birth rate had already dropped considerably. The reason for the increase in children is that women of the baby-boom generation are now giving birth and their parents have increased. Conversely in SBT, when the year class born in the age with few adults is to mature, the population will stop increase or even decrease.

Even if we begin to protect the immature fish, the adult fish will not increase immediately. It takes roughly the age at maturity. In addition, it takes more time for the survey to confirm its increase. It was in 1997 that increased control in 1989 indicated that mature individuals had begun to grow. Furthermore, as mentioned above, it does not necessarily continue to increase steadily thereafter. It takes a few generations for such a fluctuation to subside. Effects due to the intense overfishing over the past three generations will continue for a few generations (Fig. 8.5).

The Government of Japan addresses that overfishing of fisheries resources is unlikely to be extinct and fisheries resources should not be targeted by the CITES. But if stock continues to decline at the same rate as the past decline, the extinction risks a century later are not ignored. It was probably more effective to restrict imports from Taiwan and other countries that did not participate in the CCSBT by listing SBT in CITES Appendix II. Certainly, tuna fishers would be tough, but now there are surpluses of tunas in the market. SBT is often sold at supermarkets at low prices. We will be able to use SBT if we manage it appropriately. But it is not unconditional. Hundreds of millions of North American pigeons in North America quickly became extinct due to overexploitation and habitat loss (Bucher 1992). There are examples of extinction due to overexploitation. We need adequate fisheries management.

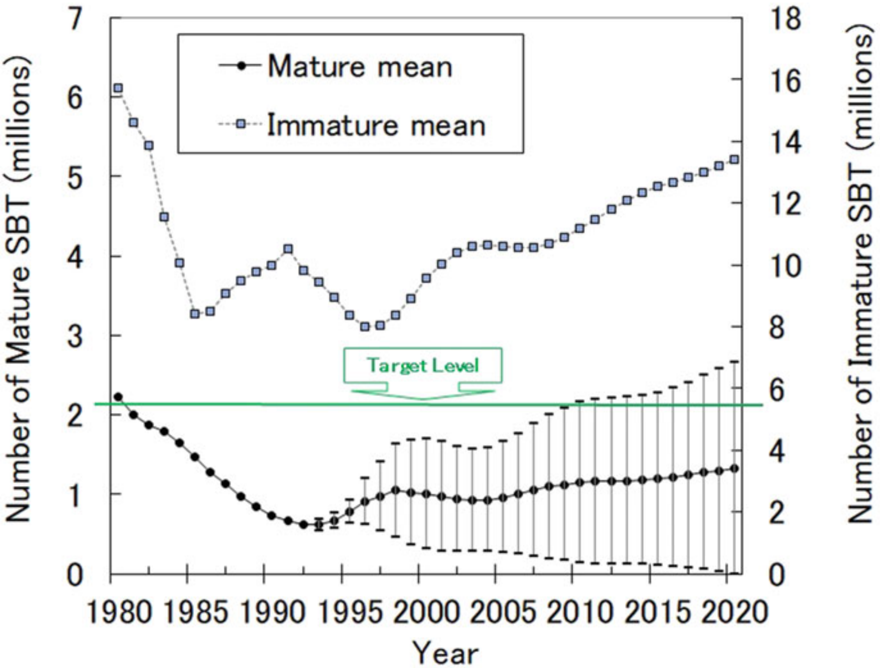

Fig. 8.5 Fluctuation of the SBT population from 1980 to 1992 (Polacheck et al. 1997) and projection of the SBT population by age-structured matrix model under past fishing mortality rate F. Lines from 1993 represent the mean numbers of immature and mature SBT from 100,000 runs of simulation. The dotted line of mature SBT represents the upper and lower 97.5% confidence limit (Mori et al. 2001)

8.4 Growth and Recruitment Overfishing

Many species of fisheries resources have multiple reproductive cycles and grow after the age at maturity, which are respectively called iteroparity and indeterminate growth. The opposite concepts are semelparity and determinate growth, respectively. This is probably related to bet hedging strategy in fluctuating environment (Katsukawa et al. 2002). The same catch yields different resource impacts, depending on the age composition of the catch. In this section, we will consider optimal fishing policy depending on the fishing coefficient by age.

There are two types of overfishing or overexploitation of biological resources: growth overfishing and recruitment overfishing. The first is the over-capture of immature and the opportunity to leave the next generation. The second occurs when fish are harvested at a smaller size than the maximum yield achieved from the cohort. We introduce the theory to understand them in a unified manner and achieve sustainable biological resource use. At the same time, we present a methodology that maximizes the economic benefits under the constraints of various environmental conditions. However, we do not consider the process error of the recruitment rate in this section.

We suppose that the survival rate of 100 g of immature fish to 1 kg of adult fish is 20%. 100 g of fish, if a fisher does not catch it, will be expected to contribute to 200 g of catch. Moreover, we also expect that the fish will contribute to spawning for future generations.

As for the relationship between development and environmental protection, we seek measures to improve the ratio of the industrial benefit (or benefit to society) of the action to the environmental loss (or risk of the loss). As long as humans exist, it is impossible to eliminate any impacts on the environment. We seek measures to reduce losses and increase benefits as much as possible. In the fishery as well, we should consider a policy to increase the catch and reduce the decrease in the number of spawning eggs left for the next generation.

To that end, it is effective to protect the immature fish and catch the adult fish. However, in the chub mackerel fishery, a large amount of immature fish was caught in the 1990s. In the 1980s, the purse seine fishery caught a large number of immature fish, while the dip net fishery caught the adult fish in the spawning season. Harvesting mature fish during the spawning season deprives the opportunity of leaving offspring of the next generation, but it is inefficient to harvest juvenile fish. We made a mathematical comparison between the impacts of these two.

We consider a cohort born in a certain year. Fish first comes to the fishing ground at the age of t_0 and set the physiological longevity to t_∞. Consider an age-specific catch policy that maximizes the yield from the entire cohort until death. We here assume that time is a continuous variable. The total yield Y from this cohort is given by the following formula:

$$Y = N_0 \int_{t_0}^{t_\infty} L(x)F(x)V(x)e^{-\delta x}dx \tag{8.1}$$

where N_0 means the number of individuals (eggs) at birth; $F(t)$, $M(t)$, $V(t)$, and $L(t)$ represent the fishing coefficient, natural mortality coefficient, profit from fishing (price or body weight) of a fish at age t, the survival rate from birth to age t, respectively; δ is the economic discounting rate, which usually takes a value of 3–5%/y. The number of fish that survive to the age of x is $N_0L(x)$, which is caught at a fishing coefficient of $F(x)$, and the value per fish is $V(x)$, but discounted by $e^{-\delta x}$ since it is x years ahead.

The survival rate is given by:

$$L(t) = g(N_0) \int_{t_0}^{t} e^{-M(x)-F(x)}dx \tag{8.2}$$

where $g(N_0)$ represent the survival rate from birth to the recruitment age t_0, which is a function of N_0. $g(N_0)$ is usually a monotonic decreasing function of N_0, i.e., $g'(N_0) < 0$. We also ignore the density effect after the age at recruitment.

We also assume that the contribution of the population to future generations left by this cohort is given by the following equation:

$$P(r) = N_0 \int_{t_0}^{t_\infty} L(x) m(x) e^{-rx} dx \tag{8.3}$$

where $m(t)$ is the number of eggs laid at age t and r is the intrinsic rate of population increase. When the population is increasing, early-born offspring grow earlier to give birth to grandchildren, which contributes more to population growth. Therefore, giving birth x years later is discounted by e^{-rx} if $r > 0$. When $r = 0$, $P(0)$ represents the total number of spawning eggs of the next generation left by the cohort. In the following, we assume $r = 0$, but it is more reasonable to consider $r > 0$ when recovering degraded resources.

Consider the sustainability condition as $P(0) = N_0$. Under this constraint, we consider a fishing policy $F(t)$ that maximizes the total catch Y. To consider the "constrained optimization problem," we consider the following Lagrangian:

$$\psi = Y + \mu(P(0) - N_0) \tag{8.4}$$

where μ is called a Lagrange undetermined multiplier. The optimal solution is obtained by using $\partial\psi/\partial F(t)$ and $\partial\psi/\partial\mu$. Since the survival rate until age t depends on the fishing policy $F(x)$ for $x < t$, i.e., $\partial L(t)/\partial F(x) = -L(x)$ for $x < t$ and $\partial L(t)/\partial F(x) = 0$ for $x > t$, $\partial\psi/\partial F(t)$ is reduced to the following formula

$$\frac{\partial\psi}{\partial F(t)} = \frac{\partial Y}{\partial F(t)} + \mu\frac{\partial P}{\partial F(t)} = N_0 L(t)[V(t) - H(t) - \mu R(t)] \tag{8.5}$$

where

$$H(t) = \int_t^\infty \frac{L(x)}{L(t)} F(x) V(x) e^{-\delta(x-t)} dx \tag{8.6}$$

$$R(t) = \int_t^\infty \frac{L(x)}{L(t)} m(x) e^{-r(x-t)} dx.$$

$R(t)$ is a basic concept of ecology called reproductive value, which is the weighted sum of the number of laying eggs left from the age t until death. Similarly, $H(t)$ is the expected value of the catch that was caught at age from t to death (Matsuda et al. 1999, Matsuda and Nishimori 2003). The harvest value at the recruitment age, $H(t_0)$, is called the yield per recruit (YPR). Especially when evaluating $V(t)$ by body weight instead of fish price, the economic discount rate is often ignored and $\delta = 0$ is set. Similarly, the reproductive value at the recruitment age, $R(t_0)$, when $r = 0$ is called the spawning per recruit (SPR) defined in Chap. 7.

From (8.5) and (8.6), the following important and clear solutions are obtained:

$$\begin{aligned} F(t) &= F_{min}(t) \text{ if } V(t) < H(t) + \mu R(t), \\ F(t) &= F_{max}(t) \text{ if } V(t) > H(t) + \mu R(t), \\ F_{min}(t) \leq F(t) &\leq F_{max}(t) \text{ if } V(t) = H(t) + \mu R(t), \end{aligned} \tag{8.7}$$

where $F_{min}(t)$ and $F_{max}(t)$ represent the lower and upper limit of $F(t)$. There are limits to the measures that can be taken due to technical constraints.

When a fisher encounters a chance to catch a fish of age t, the expected value of the profit from future catch of the fish after growing may be larger than the present fish price $V(t)$. If the fish will be caught at age x, the expected profit is $V(x)[L(x)/L(t)]$ with probability $F(x)$. In addition, the contribution to next-generation resources represent $\mu R(t)$. If the sum $H(t) + \mu R(t)$ is larger than $V(t)$, it is better not to catch, and vice versa.

However, we must decide the value of undetermined multiplier μ and the values of $H(t)$ and $R(t)$ depend on the fishing coefficient $F(x)$ for $x > t$. Therefore, the optimum solution, denoted by $F^*(t)$, is calculated for time step Δt in the following.

First, the value of μ is temporarily decided. At the final age T, $F^*(t) = F_{max}(T)$ is obvious. Since we obtained $F^*(T)$, $H(T - \Delta t)$ and $R(T - \Delta t)$ are determined and we obtain $F^*(T - \Delta t)$. If $F^*(x)$ is determined for $x \geq t + \Delta t$, we obtained $H(t)$ and $R(t)$ and therefore we determine $F^*(t)$ by backward calculation. We also calculate $P(0)$ by using $F^*(t)$ for $t_0 \leq t < T$.

We suppose two cases about $g(N_0)$. First, we suppose $g(N_0)$ is constant. If $P(0) < N_0$ for an interim value of μ, μ should decrease to satisfy the sustainability constraint. If $P(0) > N_0$, we can increase μ. We can find the value of μ that satisfies $P(0) = N_0$, as shown in the Excel file uploaded on the website. Second, we suppose $g(N_0)$ is a decreasing function of N_0. We can find N_0 that satisfies $P(0) = N_0$, the total yield Y is given by a function of μ. We can find optimal μ to maximize Y.

We show the case of Pacific bluefin tuna (*Thunnus orientalis*). The tuna was heavily caught by purse seine fishery in the summer of the spawning season when the fish price is low. It was also caught by the longline fishery and the hooking fishery in the winter season when the fish price is twice higher than in the summer. The bluefin tuna matures at about 30 kg at the age of 3 but continues to grow until it reaches about 200 kg at the age 12 years We assume that the fecundity increases in proportion to body weight at age 3 and older.

The fork length of the bluefin tuna, denoted by $l(t)$, follows the von Bertalanffy's growth curve:

$$l(t) = l_\infty \left[1 - e^{-\kappa(t+\tau)}\right] \tag{8.8}$$

where $(l_\infty, \kappa, \tau) = (249.6 \text{ cm}, \ 0.173, \ 0.254 \text{ year})$ (Shimose et al. 2009). This formula is theoretically derived from the following differential equation and is often used in growth curve:

$$\frac{dl}{dt} = \kappa(l_\infty - l) \tag{8.9}$$

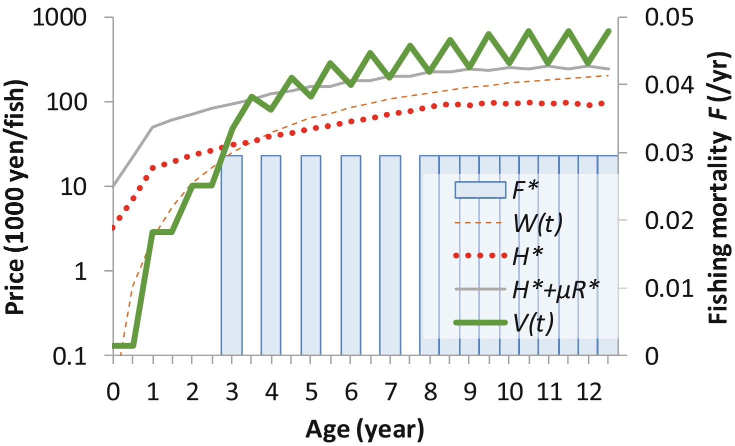

Fig. 8.6 Seasonal fish price $V(t)$, optimal fishing policy $F^*(t)$, harvest value $H^*(t)$, and reproductive value $R^*(t)$ of Pacific bluefin tuna (Matsuda et al. 2020)

The relationship between the body weight denoted by $W(t)$ and body length $l(t)$ can often be approximated to a linear relationship using a log-log graph. This is called the Allometry relationship. The weight should be proportional to the cube of body length if it grows in a similar manner, but in the case of bluefin tuna it is approximated by the following regression relationship.

$$W(t) = W_0 l(t)^{\alpha} \tag{8.10}$$

where $(W_0, \alpha) = (3.32\text{ kg}, 2.89)$ (Shimose et al. 2009). The natural mortality $M(t)$ were 1.6/year, 0.386/year, and 0.25/year at 0, 1, and 2 years, respectively. We assumed $t_0 = 0$, $\Delta t = 0.5$, and $T = 21$ years Since the initial survival rate $g(N_0)$ is unknown, here we set the % SPR = 40% as the sustainability condition to be described in Chap. 7. From these assumptions and fish price in summer and winter at each age is given in Fig. 8.6, we obtained the optimal fishing policy when $\mu = 2300$ yen/kg. As is seen by comparing the fish price $V(t)$ and $H^*(t) + \mu R^*(t)$ shown in Fig. 8.6, the tuna should not be caught until the age of 3 years. Until the age of 9 years, the fish should be caught only in the winter season and not in the summer when the fish price is low. For the age 10 years and older, the fish should be caught throughout the year. We also note that $V(t) < H^*(t)$ for the age at 2.5 years, which means that catch under 2.5 years old is also inefficient in terms of catch from the cohort without considering sustainability. Catching fish at an age when the fish price is lower than the harvest value is called growth overfishing.

The reproductive value increases with age, and in fish with an indeterminant growth, the number of spawning eggs continues to increase even after the age at maturity, so the reproductive value also continues to increase and tends to decrease near to the physiological longevity. It is peaked just before the spawning season and hits the bottom just after the spawning season. If the mortality coefficient is about

25%/year, the range of seasonal fluctuation of reproductive value is less than 25%, which is not remarkably different. In the case of bluefin tuna, the meaning of protecting the adult fish during the spawning season is mainly because of the seasonal difference in fish price rather than the difference in reproductive values. In any case, overfishing is continued in the summer because different fishers caught in the summer and winter.

Harvest value is monetary value, but the reproductive value is not. In this way, in constrained optimization, indices with different dimensions are compared. The Lagrange multiplier μ translates the reproductive value into monetary value. This method could be applied when considering various environmental problems as constraints, such as cap and trade system in greenhouse gas emissions.

Furthermore, we can draw a similar figure using the actual fishing coefficient by age and μ of the optimum catch policy (Matsuda et al. 2020).

In this way, we can formulate various environmental conservation measures in the framework of environmental economics if the upper limit of each type of environmental impact is set as a constraint condition. Also, the more difficult it is to satisfy the constraint, the more the shadow price μ will rise, and the profit Y will be sacrificed to satisfy it. In a similar way, multiple environmental impacts can be included as multiple constraints.

When discussing ecological risk, the magnitude of environmental impact cannot be determined deterministically. However, if the expected value can be evaluated, the same formulation as above is possible if social constraints are imposed on the total expected value of environmental impacts. In this way, different types of ecological risk can be compared.

However, it is unsure whether it will be a socially agreeable solution. The above is solved under the condition that each constraint is given independently of the utility, and when there are multiple constraints, they are given independently and satisfy each other. Therefore, one constraint can be satisfied with sufficient allowance, but the other can satisfy the former and the latter independently even when they are extremely strict. In addition, there may be cases where there are too tight restrictions on profits, but this means that weigh profits and environmental impacts. If we seek the policy to optimize the profit under some constraints, it would be better to convert the environmental impact into a cost and maximize the utility.

Furthermore, this method does not take into account the uneven distribution of benefits and burdens that are commonly found in economics. Even in the above cases, the act of protecting the impact on the fisheries resource and the one that receives economic profits are not generally the same. Not everyone enjoys the benefits of protecting the environment. When discussing this inequality in allocation of burdens, the value of the environment would have to be evaluated as the interests of individual stakeholders. The individual quota (IQ) system, which allocates the available catch quota to each fisher, is seen as a rational means of preventing growth overfishing. However, fisheries have different fish prices and impacts on the resource between seasons, and the harvest and reproductive values of each fishery are not taken into account when allocating IQ.

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Part III
Wildlife Management and Conservation

Chapter 9
Red List of Japanese Vascular Plants and Environmental Impact Assessment

Hiroyuki Matsuda and Kazuo Muneda

Abstract In order to create the Red List of Japanese Vascular Plants, more than 500 taxonomists, including amateurs, collaborated to collect plant distribution and evaluated various extinction risks based on the rate of population decrease. As a result, it was suggested that there are species with a high risk of extinction if the rate of decrease is high, even if the species are currently widely distributed and have a large number of individuals. Conversely, some species have a high rate of decline but a low risk of extinction.

We proposed a method to utilize this database for environmental impact assessment. At the 2005 Japan International Exposition held in Aichi, we compared the numbers of threatened species with and without the project and compared the magnitude of extinction risk nationwide. Although there is no clear threshold for the impact on biodiversity in Japan guideline for environmental impact assessment procedures, it is possible to compare the amount of increase in extinction risk by comparing it with past conservation measures and actual impacts elsewhere. In order to compare the magnitude of the impacts on various species, we proposed an index called "expected loss of biodiversity" (ELB), which is weighted by the length of the phylogenetic tree unique to the species from related species.

Keywords Extinction risk · Threatened species · Expected loss of biodiversity · Vascular plant · Red list criteria · Demographic stochasticity

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9.1 About 7000 Taxa of Vascular Plants and 400 Investigators

In 1997, the Environment Agency, Japan, published the Red List of vascular plants in Japan. This is based on plant species distribution in 4400 grids of circa 10×10 km in Japan. About 400 plant taxonomists answered the species names, the number of individuals, and the rate of decline in each grid. An unprecedented, large-scale survey of the world has revealed that bellflower (*Platycodon grandiflorum*) and thoroughwort (*Eupatorium fortunei*) were threatened species. These herbs are included in the traditional seven autumn herbs in Japan, in which the other five herbs are bush clover (*Lespedeza bicolor*), pueraria (*Pueraria lobata*), maiden silvergrass (*Miscanthus sinensis*), dianthus (*Dianthus auperbus*), and patrinia (*Patrinia scabiosaefolia*). This is "the world's first quantitative projection of plant species loss at a national level, with stochastic simulations based on the results of population censuses of 1618 threatened plant taxa in 3574 map cells of ca. 100 km^2" (Kadoya et al. 2014). We will introduce the details of this process.

In 1989, the Japanese Society for Plant Systematics (JSPS) and the Nature Conservation Association of Japan (NACS-J), a non-governmental organization, created the Japanese Plant Red List. According to this, about 1/6 of the Japanese vascular plants (spermatophytes and ferns) were listed. In 1994, IUCN completely revised the IUCN Red List Criteria (see Chap. 8). In response to this revision, the JSPS has revised the plant Red List and conducted a large-scale survey nationwide to collect the information necessary to apply the new criteria (ver. 3.0). Thus, in 1997, the Red List of vascular plants corresponding to the criteria (ver. 3.0) was published. This time, the Society of Plant Taxonomy was commissioned by the former Environment Agency of Japan.

Excluding horticultural varieties and exotic species, there are about 7000 taxa (species and subspecies) of vascular plants in Japan. Among them, the JSPS surveyed 2000 taxa that are suspicious to be threatened. The land of Japan is divided into about 4400 grids in circa 10×10 km maps. On each grid, we surveyed how many individuals and habitats of the 2000 taxa of vascular plants were, how much decreased compared to 10 years ago, and what caused the decrease. The number of individuals is the number of digits (0, <10, <100, <1000, >1000), and the rate of decrease is "not decreased," "0– 50%," "50– 90%," "90– 99%," "$>99\%$," or "local extinction" or "unknown." The JSPS asked for about 400 investigators on a questionnaire. The cause of the decrease was selected from items such as "harvesting" and "land development."

Unfortunately, the basic information in the Plant Red Data Book does not include enough information on the seagrass in aquatic ecosystems. It was difficult for 400 investigators to search from the mountains to the sea. Although the species identification is insufficient, the Environmental Agency survey has detailed examined the loss rate of tidal flats, seagrass/seaweed beds, and coral reefs. This basic information could be very helpful in protecting coastal ecosystems.

Table 9.1 Number of grids distributed by population and decline rate of primura (Yahara et al. 1998)

	Population decline rate per decade						
Population size	<0.01	<0.1	<0.5	<1	>1	Unknown	Total
>1000	0	2	0	1	1	4	8
>100	2	2	1	3	2	5	15
>10	5	16	19	6	2	12	60
>1	1	3	3	2	1	2	12
Unknown	−13	0	1	0	0	22	23
Total	8 + 13	23	24	12	6	45	118
%	16.1%	9.2%	26.4%	27.6%	13.8%	6.9%	100%

The survey was entrusted to about 400 experts who are accurately able to identify any species or subspecies of the local plant. They were not only university faculties, but also museum curators and high school teachers, and surveyed as volunteers. Since the method of survey is different from that of the Plant Red List in 1989, the population of 10 years ago was not exactly known. The rate of decrease depends on the subjective judgment of the investigators. Because of the choice of unknown population and rate of decline, some respondents frequently chose as unknown and the others intuitively chose some rank. It would have been a tremendous effort for 400 investigators to go all the way through mountains, wetlands, and coastal vegetation. In addition, it seems that the task of totaling the answer sheets is also difficult. A total of ca.2000 taxa × 4400 answer sheets (in fact, there are no such answers from places where they are not distributed, so it is much less than this) was input into a spreadsheet file and analyzed. An enormous amount of work was done by an associate professor and students at Niigata University.

In this way, the tabulated results shown in Table 9.1 were compiled for each taxon. This is an example of *Primula sieboldii*. There were 19 grids where the population size is between 10 and 100 and the population decline rate is between 10% and 50% for the past 10 years. There are 13 grids where this species had once existed but went locally extinct. We ignored grids with unknown population sides in the estimation of nationwide population size. In other words, it is still distributed in more than 95 grids, but 44 grids have been reduced to less than 1/10 of the population 10 years ago, including extinct areas. This accounts for >50% of the 118–45 + 13 = 86 grids whose decline rates were reported.

9.2 Red List of Japanese Vascular Plant

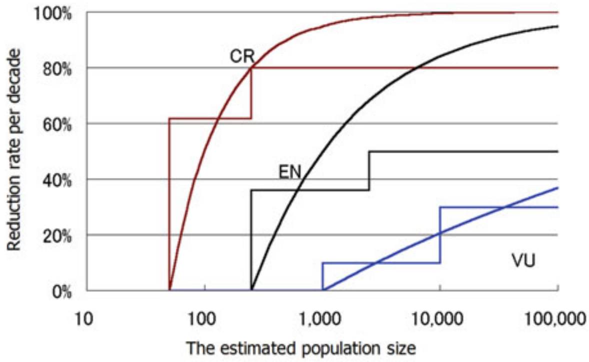
There were many taxa that had no record at all. These must be treated as data deficient (DD). The data shown in Table 9.1 was collected for about 1500 taxa. We made a Japanese Red List based on this database. We cooperated in the development of the method. In making the method, we kept the following in mind. First, we developed the method to judge the Red List category of each taxon from the published information. All the above data for threatened taxa will be published in the Plant Red Data Book published by the Environment Agency of Japan in 2000. Second, we publish the analysis method including the source code. Although it is not completely the same as the method adopted by the taxonomic society described below, the method we proposed has the source code of the computer program C language published on my website. Third, even taxon with limited data can be evaluated. Fourth, it should conform to the IUCN 1994 criteria as closely as possible. Finally, we preferred a relatively simple method.

First, the current population size is estimated from the data in Table 9.1, although the exact number is unknown, and the digit of population size might be correct. Therefore, several individuals were considered as the geometric mean of 1 to 10, i.e., $\sqrt{10} = 3.16$ individuals. In similar ways, several, dozens, hundreds, and thousands of individuals were considered 3.16, 31.6, 316.2, and 3162.2, respectively. Since there are 12, 60, 15, and 8 regions, respectively in the case of primula, there are a total of 31,197 individuals! Of course, this is not an exact number and should be considered tens of thousands.

Now, starting from this current state, we estimate the future population size. Consider the initial distribution where the above population exists in each region. We assumed that by 10 years, each region will decrease according to the abovementioned frequency distribution of nationwide population decline rate. As for the 13 extinct grids (the reason for adding one region will be explained later), the number of individuals 10 years ago is unknown, so we assumed the reduction rate of 99% in these grids. That is, the reduction rate is >99% or more, 90–99%, 50–90%, and 0–50%, and 0% with the probability of 0.161 + 0.092, 0.264, 0.276, 0.138, and 0.069, respectively. This does not mean that the rate of decline in the grid from 10 years ago will continue in the future, but rather randomly selects a decline rate from the distribution of the rate of decline throughout the nation and assumes that the population in each grid will decrease. This is based on the premise that the same region is not always developed but developed nationwide. In addition, if a decrease of 90–99% occurs, we assumed that the reduction rate is selected from uniform random numbers between 90% and 99%.

When computer experiments are continued in this way, the number of individuals in each grid gradually decreases or disappears. If the number decreases to less than one individual, the grid is considered locally extinct. We consider the taxon to be extinct when it disappears from all grids. This computer experiment is repeated 10,000 times. Depending on the generated random numbers, they may be extinct sooner or may last longer. If the frequency of extinctions by 100 years is 1000 times among the 10,000 trials, the probability of extinction after 100 years is 10%. In this way, we calculate the extinction probability $P(t)$ from 10 years to 300 years. Since the calculation is performed in 10-year units, if a person survives t years later and becomes extinct t + 10 years later, it is considered extinct t + 5 years later. The average time to extinction T is calculated by

Fig. 9.1 The relationship between the ACD criterion (curve) used in the Japanese Vascular Plant Red List and the IUCN criterion (polyline) (Yahara et al. 1998)



$$T = \sum_{i=1}^{30} (10i + 5)[1 - P(10i)] \tag{9.1}$$

Assuming that the average value of the total number of individuals after t years is $N(t)$, the reduction rate 1-R is obtained as 1-R = 1-$N(10)/N(0)$.

The extinction probability obtained in this way is applied to criterion E. In other words, if the extinction risks exceed 50% after 10 years, exceeds 20% after 20 years, and exceeds 10% after 100 years are CR, EN, and VU, respectively. In addition, in order to judge from the decreasing trends of the population, the "ACD" criterion, which is a combination of the IUCN's criteria A1, C1, and D1, was set. In order to overcome the "defect" that criterion A does not reflect the population size, criterion "ACD" refers to the current population size N_0 and the population decline rate 1-R determined above. Under the assumption that the population continues to decrease at the rate 1-R, if the population becomes <50 after 10 years (i.e., $N_0(1\text{-}R) < 50$), <250 after 25 years ($N_0(1\text{-}R)^{2.5} < 250$), and <1000 after 100 years ($N_0(1\text{-}R)^{10} < 1000$), the species was ranked as CR, EN, and VU, respectively. In this criterion, the boundary of each category is represented by a curve of population and reduction rate as shown in Fig. 9.1, which is smarter than IUCN's criteria A, C, and D, which make zigzag lines.

The above methods for estimating populations, decline rates, and extinction risks used in the Red List of Japanese vascular plants are a matter of convenience. The Japanese Red List in 1989, which relied on expert judgment, has been changed to a more objective and quantitative method. However, the use of common techniques to analyze about 1500 plants is unprecedented in the world. This was possible because there were 400 investigators who could accurately identify vascular plants. It is important that endangered species survive and that investigator skills be passed on to the next generation.

The calculation result of the extinction risk is shown in Fig. 9.2. Widely distributed species have a large population and are not likely to be extinct immediately. If their decline rate is high, the extinction risk will rise over several decades. In the end, more than 1400 taxa were designated as threatened. It accounts for 20% of 7000 taxa

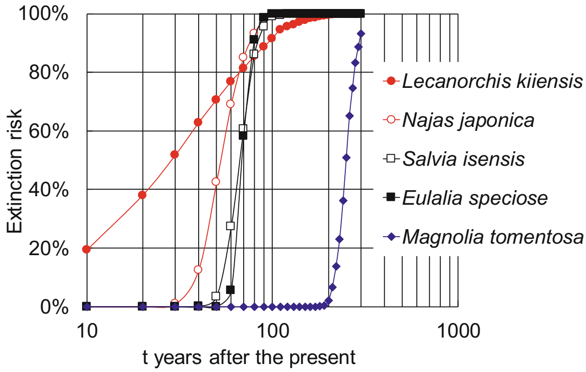


Fig. 9.2 Extinction risks of several plants, *Salvia isensis* and *Eulalia speciose*, were determined to be VU by criterion E, and *Lecanorchis kiiensis*, *Najas japonica*, and star magnolia (*Magnolia tomentosa*) were determined to be CR, EN, and VU by the ACD criterion, respectively

of Japanese vascular plants. With this listing, we cannot protect all threatened species. However, the detailed information on the population, distribution area, and reduction rate provide a quantitative evaluation on the process of species disappearance. We will describe some methods in the section of environmental impact assessment.

The biggest factor of threatened species is habitat loss due to land use change. For plants such as the orchid family, the main causes of which are overexploitation due to harvesting. JSPS does not disclose the detailed distribution information of species including orchid family that are threat by overexploitation. The Ministry of the Environment, Japan, discloses the distribution of other plants in ca.10 × 10 km grids.

About 22% of Japanese wild plants were listed in the Japanese Red List of vascular plants (Environment Agency of Japan 2000). Conservation measures are needed for these taxa, and continuous surveys are needed to compensate for the lack of information, and extinction risks need to be reevaluated with more detailed numerical data. Therefore, the JSPS (2003) decided the purpose of the survey for the revision of the RL as follows:

1. To accurately understand the current status of CR species.
2. To evaluate taxa that have leaked from the previous survey and are suspected to be CR.
3. To investigate areas that were insufficiently surveyed in the previous survey (especially remote islands).

In addition, it was decided to confirm the population of EN and VU species. The purpose of the revision work is to reevaluate the status of each taxon based on the data obtained from the review survey and increase the reliability of the Red List.

In the previous survey, all 2100 taxa were surveyed with the same effort and surveyed the same items for each taxon. However, the review survey focused efforts on taxa of CR or with insufficient information. In remote islands, the survey unit (grid size) has also been changed from 100 km^2 to 1 km^2.

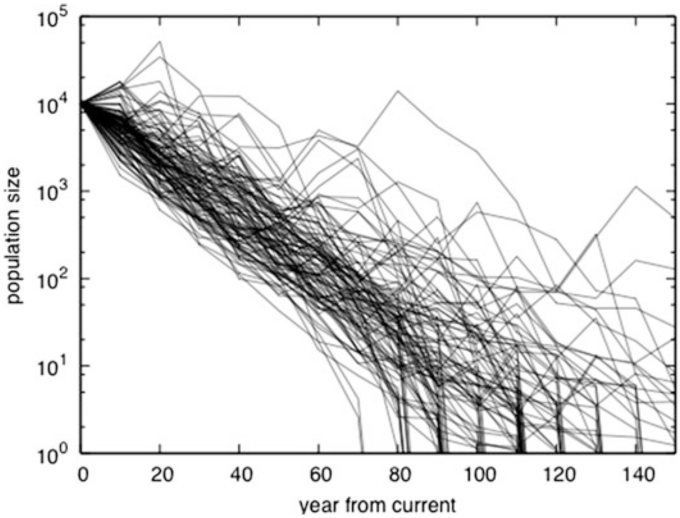


Fig. 9.3 Examples of population dynamics; 100 simulation results are illustrated. The horizontal axis represents the elapsed time from the present and the vertical axis represents the number of individuals (logarithm scale)

A simulation was performed to calculate the extinction risk of 1883 taxa among the 1926 surveyed species.

In the review survey, the reduction rate of the number of individuals for each grid was estimated from a comparison between the local population size in the previous survey and in the review survey. We considered the probability that the population increased in the grid, unlike the previous analysis. As a result, the computer experiment was as follows (Fig. 9.3).

9.3 Demographic Stochasticity and Environmental Stochasticity

There are two major factors of species extinction. One is, as explained in Chap. 7, that if the population and habitat continue to decrease consistently from the past to the future due to human impacts. The other is that a species, which have been heavily declined but is not impacted now, may go extinct by chance. Even when it is decreasing, when it becomes extinct is a matter of stochasticity. Conversely, human is not innocent, even if they were extinct directly without human intervention. Originally, if there were more populations and lived in many habitats, they would not have been extinct.

Of the five IUCN criteria, criteria B and D are criteria for the danger of extinction of organisms whose populations and distribution have been sufficiently reduced. In the criterion B, the area of occupancy (AOO) is defined as the area where organisms

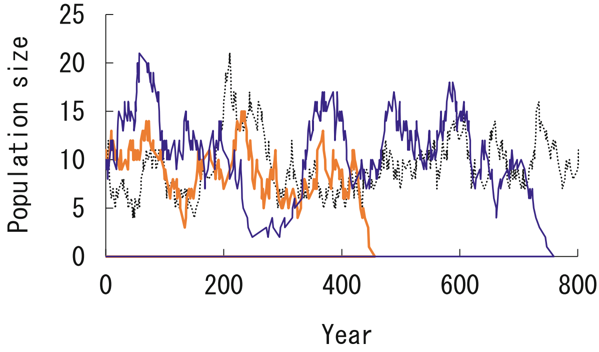


Fig. 9.4 Examples of population fluctuation due to demographic stochasticity. It goes extinct after about 456 years and 760 years in the unfortunate cases, while the grey case persisted for >800 years. These 3 simulations fluctuated around the 10 individuals that are the carrying capacity

actually inhabit. This is difficult to measure. So, as a practical concept, we consider the extent of occurrence (EOO), which is defined as a convex figure surrounding the habitat. For example, we consider a 10 × 10 km grid that includes the habitat as EOO. If we use smaller grids of 1 × 1 km, the EOO is evaluated as narrower. If a sufficiently fine-grained map is used, it can be considered the AOO. How to define it is ambiguous and left to the user. The method should be clearly described at the stage of use.

There are two types of stochasticity in the extinction processes. The first is environmental stochasticity. Even if the number decreases, the population persists under good environmental condition but extinguishes under bad conditions. If the quality of the environment falls on all individuals equally, the magnitude of environmental stochasticity increases in proportion to the population.

Another is demographic stochasticity. While the population is large, the number of recruit close to the average. But as the population decreases, the number of recruit is actually left by chance. For example, it should survive with a probability of 50%, but if there are only 10 individuals, the probability of annihilation is $(1/2)^{10} = 1/1024$, and the probability of exactly 5 individuals surviving is 24.6%, given by binomial distribution. Seventy-five percent are more than 6 or less than 4. If the population is large, lucky mothers and bad ones will be mixed and the fluctuations will be offset, so the magnitude of the demographic fluctuations is proportional to the square root of the population size. This is the same as the relationship between signal and noise (S/N ratio). When the population is less than several tens, the risk of extinction cannot be ignored even with demographic stochasticity (Fig. 9.4).

9.4 Population Viability Analysis

The extinction probability can be theoretically calculated for a species whose population, decrease rate and its annual variation are known, assuming that the declining trend will continue in the future. This is called Population Viability Analysis. Whether the whole population is grouped together or divided into subpopulations greatly affects the risk of extinction. If this is ignored, the basic equation of extinction probability becomes

$$\frac{dN}{dt} = r(N)N + \sigma_e Z_e(t) \circ N + Z_d(t)\sqrt{N} \tag{9.2}$$

(Hakoyama and Iwasa 2000), where $N(t)$ is the number of individuals in generation t; $r(N)$ is the rate of increase per individual and is a function of N when density effects exist. For example, we assume the logistic growth $r(N) = r_{\max}(1\text{-}N/K)$; σ_e is the magnitude of environmental fluctuations; $Z_e(t)$ is a random number representing environmental fluctuations; and $Z_d(t)$ is a random number representing demographic stochasticity. The random number is given by a random variable called white noise. In this formula, the unit of t is not 1 year, but the average generation time. However, long-lived organisms take much time to extinction, but preservation measures are too late. In the Japanese vascular plant Red List, the time unit is not referred to as generation length, but to physical years. This corresponds to the ACD criterion as shown in Fig. 9.1. As mentioned above, environmental stochasticity is proportional to the population size N and demographic stochasticity is proportional to $\sqrt{N}$. When the population fluctuates around the carrying capacity K, $r_{\max}$, and σ_e are estimated from the time series of past population fluctuations. However, the estimation error is large (Hakoyama and Iwasa 2000).

Criterion D1 is a criterion determined based on such a theory. The 50 individuals, which are the criteria for CR, are at a high risk of extinction, considering only demographic fluctuations, and are called the minimum viable population size (MVP). In order not to lose genetic diversity, at least about 500 individuals are required, which is called genetic MVP. The above population is distinguished from this and is sometimes referred to as demographic MVP. The populations of 50 and 500 are only reference points. In addition, the risk of extinction cannot be ignored even if the population is larger than MVP, not a criterion that there is no danger of extinction.

For species where the population is continuously decreasing and the density effect can be ignored, the extinction probability is calculated from the current population $N(0)$, the average r^* and the standard deviation $\sigma_e^{\,2}$, and the autocorrelation ρ. In this case, when extinction depends on demographic fluctuations, but extinction must happen sooner or later. If the average decline rate is large, the time to extinction can be reduced even if the demographic stochasticity is ignored (Lande and Orzack 1988). The probability $G(t)$ that the logarithm of the number of individuals ($x(t) = \log N(t)$) at least once becomes less than x_c is (9.2)

$$G(t) = \frac{(x_0 - x_c)}{\sqrt{2\pi\sigma^2 t^3}} \exp\left[-\frac{(x_0 + r^* t - x_c)^2}{2\sigma^2 t}\right], \tag{9.3}$$

where

$$\sigma^2 = {\sigma_r}^2\left[1 + 2\sum_{\tau=1}^{\infty}\rho(\tau)\right].$$

The extinction risk is obtained if $x_c = 0$, but demographic stochasticity should be taken into account.

As is seen from this Eq. (9.2), the extinction probability depends on the current population x0, the rate of decline r*, and its variation sr. If these data are available from past time series of population trends, criterion E can be used. Criterion E is called quantitative analysis and is introduced in the Japanese plant Red List. Population viability analysis is one of them, and it is necessary to write down what data and assumptions were used.

Furthermore, if the organism is divided into several subgroups, or if there are local movements and some movement between them, further detailed analysis considering the metagroup is necessary.

As explained in the previous section, if the extinction risk cannot be evaluated directly, the four criteria A–D in Table 8.1 are useful. Criterion D1 is a criterion based on only the current number of mature individuals. It is based on the minimum viable population (MVP). Criterion A is a criterion in which the absolute value of the number of individuals is unknown, but is determined only based on the rate of decrease compared to a certain time in the past and the present.

There is not always a clear evidence for these quantities. Experts made decisions at IUCN workshops. However, it is not without grounds at all. For example, a CR reduction rate of 80% can be interpreted as follows. In the case of 250 or more individuals that do not meet the other criteria A–D, if they are reduced by 80% in 10 years or 3 generations, they fall CR under criterion D1.

According to the 1994 criteria, any one of the five criteria for A–E in Table 8.1 should be met. A species that satisfies any of criteria A–D is listed even though it does not meet criterion E, or its extinction risk is low. If the population is divided and there is a risk of a sudden disappearance, criterion E may be satisfied even if criteria A–D are not met. However, in most cases, satisfying E will satisfy other A–D criteria.

It is a big attempt to create objective criteria and evaluate all multicellular organisms in a unified manner. It is reasonable that it is linked to extinction risk, albeit incomplete. It is still in the process of development, and future research on the Red List will continue to contribute to the protection of biodiversity.

9.5 Application to Environmental Impact Assessment

Environmental impact assessment (EIA) is defined as “the process of identifying, predicting, evaluating and mitigating the biophysical, social, and other relevant effects of development proposals prior to major decisions being taken and commitments made” (IAIA 1999).

Depending on the country, the EIA goes through the following steps. First, the proposer of land development or another will deliver a methodology document that describes the project plan, EIA evaluation items, and evaluation methods. Relevant government agencies, local governments, and citizens respond to this as public comment. The proposer investigates the environmental impact after modifying the method based on these comments. Based on the survey results, the proponent will revise the plan to avoid or reduce environmental impact as much as reasonably achievable. They may propose to take compensation measures outside the project area for the impact that cannot be reduced within the area. These are put together in a draft environmental impact statement. Stakeholders have the second chance to give comments. Based on this opinion, a revised plan will be published as an environmental impact statement. If the proposer makes significant changes to the plan, return to the methodology document or the draft EIS stage.

In Japan, the Environmental Impact Assessment Law, which regulates EIA procedures, was put into effect in 1999. Authorization of the business is done by another law. Comments at the process of EIA can be referred to for the authorization, but the EIA itself is not an examination by a third party, but a process for consensus building between the business proposer and local stakeholders.

Although the Red List stated in Chap. 8 that it was not legally binding, impacts on threatened species are to be assessed in the EIA, and conservation of their habitats will be considered. However, there is no standard on how much the risk should be reduced. According to the Basic Matters related to EIA in Japan, "The evaluation is to be conducted, based on the results of survey and forecast, by *clarifying the project proponent's view* on whether the project proponent has avoided or reduced, within the possible limits, the impact on environmental components relating to the selected items likely to be caused by the implementation of a target project" (MOE 1997).

Just before the enforcement of the EIA Law in Japan, the 2005 World Exposition "Aichi Expo" held in Seto City and Nagakute Town, Aichi Prefecture, was advocated as a precedent case for Japan's EIA system. The Red List and Red Data Book (RDB) of Japanese Vascular Plants based on extinction risk were published in 1997 and 2000, respectively. Using the number of grids by population size class and the frequency distribution of the rate of decline by grids for each endangered species described in the RDB, we can calculate the extinction risk of the endangered species in 2000. During the development of the Red List of Japanese vascular plants, we considered how to apply the database for the Red List of Japanese vascular plants to EIA.

When the nationwide population size of a threatened species or subspecies is N_p, the number of extant grids is L, and the population decline rate is R, the mean time to extinction, denoted by T is calculated by the data in the RDB. To this end, we have released the source code of C language that can calculate the extinction risk by the Monte Carlo method from the data published in Red Data Book. We also made the regression model for $T(N_p, R, L)$ as follows (Matsuda et al. 2003):

$$T\left(N_p, R, L\right) = A - B\frac{\log\ \left(N_p - m\Delta N\right)}{\log\ (1 - R)} + C\ \log L \tag{9.4}$$

where $(A, B, C) = (2.71, 4.65, 4.56)$. m is a coefficient representing the survey strength. At the Aichi Expo, the endemic species of this region, star magnolia (*Magnolia tomentosa*), received particular attention. In the process of the EIA related to Aichi Expo, we found that the number of star magnolia found in the Expo site described in the draft EIS was larger than the number of individuals in this area that was counted in undisclosed data used for calculation of extinction risk in the Red List. We considered that this is because the survey intensity is different. It might have been necessary to consider that the true population was underestimated in the Red List. But we assumed $m = 0.1$.

We assumed that the number of individuals of the species was reduced by ΔN_p caused by the target project, whose number is published in the EIA process. Assuming that this project is transient and does not affect the nationwide reduction rate R, the average extinction waiting time T with and without the project are $T = T$ (N_p, R, L) and $T^* = T(N_p - m\Delta N, R, L)$, respectively. Here, the difference between the reciprocals was regarded as an increase in extinction risk (Oka et al. 2001). In other words, $\Delta(1/T) = (1/T^*) - (1/T)$ is regarded as an increase in extinction risk due to development projects. Furthermore, instead of considering all species in the same weight, we proposed the concept of "expected loss of biodiversity" in which the weight was calculated by the time length B while the species were differentiated from related species, and the total sum $\sum B_i \Delta(1/T_i)$of all species was taken. This unit is the year (Oka et al. 2001).

As mentioned earlier, EIA does not have a standard for the extent to which the impact should be reduced, and the business proposer aims to reduce the impact according to the ALARA (as low as reasonably achievable) principle. At the Aichi Expo, the plan for preserving the habitat of star magnolia was changed during the attraction stage. We asked the Ministry of Economy, Trade and Industry, Japan [METI] to describe the process in the methodology document for Aichi Expo. And we evaluated how much impact on other species still remains in the draft EIS. We expected that the business proposer would avoid if there was a greater impact than the species agreed with the stakeholders at the invitation stage.

The METI, which is the competent authority, was enthusiastic about this proposal. Unfortunately, the MOE [The Ministry of the Environment, Japan] refused saying that probabilistic risk assessment does not fit with the EIA system in Japan. Since that time, the concept of risk has been used in Japan's EIA only to the health risk of chemical substances and the avian collision risk (Chap. 11).

At the Aichi Expo, the EIS was submitted in January 2000 with species that had a large risk of extinction remaining. However, the next day, a local newspaper reported that BIE (The Bureau of International Expositions) had been dissatisfied, and the plan was changed thoroughly by other than the EIA. Therefore, it has shown that the EIA did not work in the regulation of environmental impacts.

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Chapter 10
Adaptive Risk Management of Sika Deer

Hiroyuki Matsuda

Abstract We consider here a management policy for a sika deer (*Cervus nippon*) population on Hokkaido Island and Yakushima Island. Deer populations are characterized by a large intrinsic rate of population increase, no significant density effects on population growth before population crash, and a relatively simple life history. Our goals of management for the deer population are (1) to avoid irruption with severe damage to agriculture and forestry, (2) to avoid the risk of local population extinction, and (3) to maintain a sustainable yield of deer. To make a robust program on the basis of uncertain information about the deer population, we consider three levels of relative population size and four levels of hunting pressures on Hokkaido. We also take into consideration a critical level for extinction, a target level, and an irruption level. We recommend catching males if the population size is between the critical and target levels and catching more females than males if the population size is larger than the target level. The simulation results suggest that management based on sex-specific hunting is effective to diminish the annual variation in hunting yield. We also estimated the population size in Hokkaido. A generalized linear mixed model is used in this estimation. We then estimate the population from the index by evaluating the response of the known amount of harvest. We apply state space modeling to the harvest-based estimation to remove the measurement errors. We propose the use of Bayesian estimation with uniform prior-distributions as an approximation of the maximum likelihood estimation. Simultaneous estimation of absolute population size and the natural population growth rate is difficult. We need the product of these to adequate population control. The biological balance of Yakushima Island is also currently being compromised by the overpopulation of sika deer. To identify the best management practice for future implementation, we evaluated and compared the performances of six different zone-based management strategies. Under the current management scenario, the median population size of the sika deer on the island would temporarily decrease, but it would subsequently rebound. Under a scenario that allows management zones to be prioritized according to the occurrence of threatened plant species and deer

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population size, model simulations suggested that the scenario focusing on the central zone would show the best performance based on the probability of achievement of the management goal.

Keywords Sex-specific hunting · State space modeling · Generalized linear mixed model · Overabundance · Harvest control · Culling · Wildlife management

10.1 History of Overexploitation and Overabundant

It is not only fisheries resources that need population management. The deer has experienced population collapses and outbreaks. In Hokkaido, based on limited information, we started an experiment of population control of deer by hunting. It is considered to use deer as a natural resource.

Japanese sardine is known as organisms whose population fluctuate range is estimated to be >500 times. Mammals such as deer also fluctuate greatly. Yezo deer (*C. nippon yezoensis*) were used as meat (venison) for a short period of time in the Meiji era, and canned venison was exported. Both the horns and the fur had commercial value.

Deer population collapsed due to overexploitation during the Meiji era (Fig. 10.1). The number of catches decreased in geometric progression. On the way, heavy snowfall occurred in 1880 and 1882, and it was hunted from 1884 to 1901. Yezo wolves went extinct probably at the end of the nineteenth century. Deer hunting was once resumed but again banned from 1921 to the mid-1950s.

After the mid-1950s, deer population began to increase in geometric progression. The distribution channel for deer meat had disappeared. Deer devoured farms and

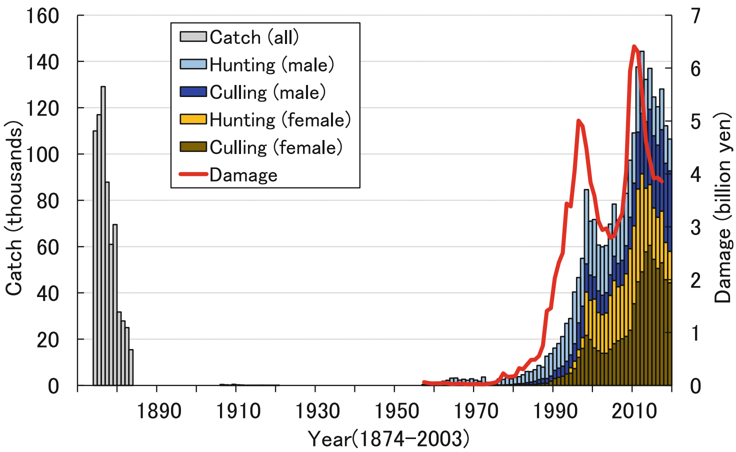

Fig. 10.1 Changes of catch in number of sika deer (drafted from Hokkaido Prefectural Government materials)

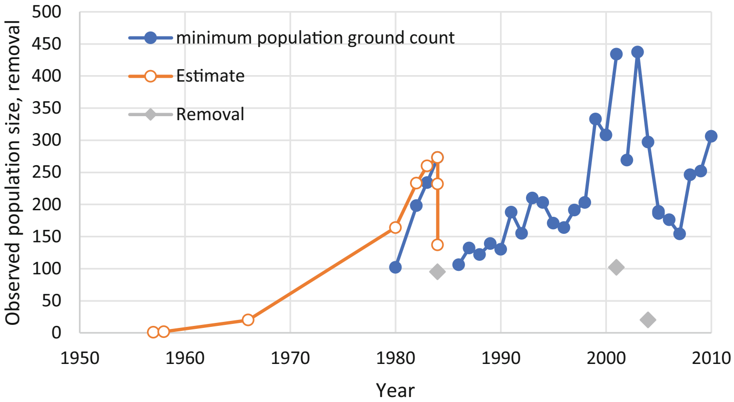

Fig. 10.2 Population changes of sika deer on Nakanoshima Island, Hokkaido, Japan, 1957–2010. Diamonds show the removal in 1983, 2001, and 2004; open circle and black circle shows estimated population size in different ways

pastures, consumed leaves in the plantation and wild plants less than 2 m high including threatened species, and strips bark. The amount of damage to agriculture and forestry, which was 1.4 billion yen in 1988, continued to increase, exceeding 4 billion yen in 1995 (Matsuda et al. 1999).

The population growth of wild animals and plants usually slows down, reaching the carrying capacity. This is called the density effect in population growth. However, the sika deer population often shows an insignificant density effect. The sika deer matures approximately at the age of 2 years and breeds one calf each year. The sex ratio at birth is approximately 1:1. It consumes many species of grass and continues to increase beyond the "carrying capacity." Eventually, a large number of males die of starvation and the population size will decrease sharply. The population collapse has been reported at Cape Shiretoko, Nakanoshima Island with Lake Toya in Hokkaido, and Mt. Kinka in Miyagi Prefecture (Kaji et al. 2005).

Three deer were released on Lake Toya within 10 years from 1957 and continued to increase at a rate of about 15% per year, reaching 299 in the fall of 1983, and 67 natural deaths by May 1984. Then, 95 deer were removed (Fig. 10.2). After the vegetation was once grazed, only grasses such as spurge (*Pachysandra terminalis*) remained on the forest floor (Kaji et al. 2005). In initial peak in autumn of 1983 was followed by a crash in winter of 1983–1984, when 67 carcasses were found and 95 deer were removed. Second irruptive event occurred to a peak in 2000 and declined in winter of 2000–2001, when 40 carcasses were found dead and 102 deer were removed to out of the island. Thereafter the population recovered to 297 deer in 2003, and crashed again (winter of 2003–2004) when 100 deer were found dead.

In the Meiji era, it is said that the sika deer died in large numbers in the heavy snowfall years. The sharp decline in the Meiji era is due to overexploitation and heavy snowfall. When a population management plan based on adaptive management was established in eastern Hokkaido in 1998, we assumed that heavy snowfall would cause mass mortality once every 20 years on average. In Hokkaido, the average temperature in winter is below freezing even in the warm winter, and it is considered that the amount of snow would not decrease.

In fact, although heavy snowfall years came in 2011, mass mortality did not occur. This is probably because the natural vegetation changed from conifer forest in the Meiji era to deciduous-coniferous mixed forest. Coniferous mixed forests are important wintering sites for sika deer (Sakuragi et al. 2003; Kaji et al. 2005).

A natural enemy, the Japanese wolves (*Canis lupus hodophilax*), have gone extinct, and due to rising hunting and animal welfare movements and the decrease in game hunters, the protection policy is so powerful that there is no threat of deer. Hokkaido has become a suitable habitat for deer due to afforestation and grassland expansion. Even if the sika deer is over-abundant, most of the deer begin to reproduce at 2 years old and breed one calf each year. Not only damage to agriculture and forestry, rail accidents and road accidents, but natural vegetation may also be degraded accross Hokkaido, like Nakanoshima island in Lake Toya.

In 2003, the Japanese Society of Plant Systematics published a statement on "Prevention of Herbivorous for Conserving Endangered Plants in Southern and Western Japan." The impact of sika deer on natural vegetation in Japan is serious (Ministry of the Environment (Japan) 2010). The population control of deer is also enforced in the World Heritage sites of Shiretoko and Yakushima.

10.2 Yezo Deer Conservation Management Policy for Eastern Hokkaido Island

In Hokkaido Prefecture, the management policy was reviewed in 1998, and the "management policy" for eastern Hokkaido population of yezo deer was implemented to keep the number of deer appropriate for humans, avoiding the threat of extinction due to overexploitation. As of 1993, the number of deer in the Eastern Hokkaido area was estimated to be 80,000–160,000. This is the value obtained by multiplying the population density and the total habitat area by counting the number of deer by the areal census. Survival rate, childbirth rate, and maturity age were estimated from a field survey in Nakanoshima Island. In incorporation of about 20% uncertainty percentage in these values, and considering the rate of natural increase as 12–18% per year, and aiming to bring it within the appropriate level after 5 years, we calculated the target number of catches.

To develop a management plan, we consider the life history of deer, mating season, breeding season, migration to a wintering location, return from a wintering location, breastfeeding, and the hunting season (Fig. 10.3). In addition to this, we need to schedule the monitoring of population count, the period for data compilation and analysis to estimate the population size, decision of target number of catch in the next year, and decision of meeting schedule to determine the length of the next hunting season (Matsuda et al. 1999). The earliest and most reliable estimation of the

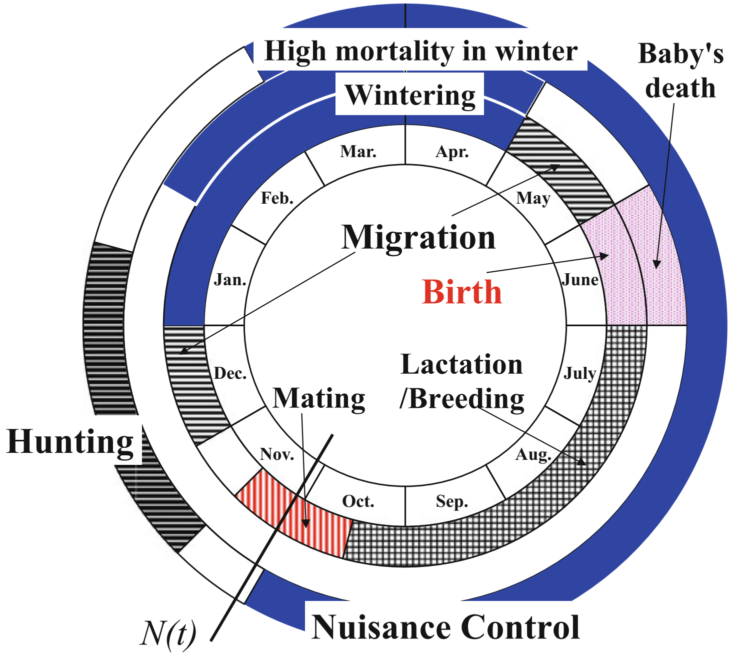

Fig. 10.3 Life history of deer (Matsuda et al. 1999). The number of individuals at the mating period (thick line) is used as the representative value for the year

population in the deer management plan is a spotlight survey conducted in the fall. Municipality staffs drive on fixed roads every year at night and light up to observe deer along the road. We estimate how much the deer is in a range on either side of the road. This is the relative index of population size. About 120 sites are surveyed every year in Hokkaido, and the population trend is read in detail. Since June is the birth period and the visual survey is autumn, the number of individuals aged 0.5 and over 1.5 can be seen. However, we do not know up to age by visual inspection, so we estimate from other data to estimate the age structure.

The management plan was based on the “feedback control” that had been discussed at the International Whaling Commission (Tanaka 1980). Since ecological information such as the population size, survival rate, and reproductive rate is not well understood, the population rends should be repeatedly monitored, protected the population when the number decreases, and removed a lot when the population is overabundant. It can be said to be the first example of adaptive wildlife management in Japan (Matsuda et al. 1999). This management system was the model for the “specific plan” introduced by the Japanese Wildlife Conservation Act, which was revised in 1999. The population dynamics model includes an annual variation of survival rate and reproductive rate, uncertainty of life history parameters (process error), and algorithm for changing hunting pressure policies according to the updated population index. This is called feedback control. Therefore, the numerical

Table 8.1 Measures for the "Conservation and Management Plan for Sika Deer" enforced in 1998

Measures	Implementation conditions	Contents of measures	Hunting female	Hunting male	Culling
Emergency reduction measures	Population index 50%[a] or more	Thorough capture (up to three consecutive years)	5	5	1
Gradual reduction measures	25% or more and less than 50%	Female deer priority capture	1[b]	1	1
Gradual increase measures	5% or more and less than 25%	Male deer priority capture	0	2	0.1
Ban on hunting measuers	less than 5% or after a heavy snowfall	Full hunting ban	0	0	0

[a]Population index means the percentage of population size in 1993
[b]For hunting and culling pressure for females and males, the strength at the gradual reduction measures is set to 1, and that for other measures is the relative value

simulations are repeated, and the risk of management failure will be evaluated. As a result of the examination, we made a management plan as shown in Table 8.1.

Four measures are used according to the population index. The population indices at the boundary are called the outbreak level, the target level, and the lower limit level, which are 50%, 25%, and 5% of the population size as of 1993, respectively.

We planned to take emergent reduction (ER) measures until the population decreased below the outbreak level. After the ER measures end within a few years, ideally we have to continue to take only gradual reduction (GR) measures and gradual increase (GI) measures except for the year following the heavy snowfall year. If it again exceeds the outbreak level or falls below the lower limit level in the future, it means management failure.

However, we cannot say that it will never fail. We cannot say that there is no risk of heavy snowfall coming for three consecutive years. There is a risk of making a large mistake in estimating the number of individuals. Therefore, we have defined that the criterion for judging management failure is once again exceeding the outbreak level or falling below the lower limit. Taking into account uncertainty in mathematical models, the target level, outbreak level, and lower limit level were set so that the risk of management failure within the next 100 years would be below the permissible limit (Matsuda et al. 1999).

In this way, we call "risk assessment" to estimate the risk of failure under a particular situation setting. Risk assessment is necessary not only for wildlife management but also for all systems with uncertainty, from financial investment to nuclear accidents. In addition, management that determine and change policies in terms of risk assessment is called risk management.

Ignoring measurement error of ecological information or assuming that environment is constant (ignoring process error) makes a big mistake. As explained in the chapter on chub mackerel fisheries, if the fish are caught in constant harvest amounts, the stock size does not stabilize. Uncertainty is inherent in the estimation.

In addition, the fact that changes in catch rates depend on population size means that fluctuations in catches are larger than fluctuations in population size. Hunters will not want a large fluctuation in the harvest. The sika deer management plan recommends catching a large number of females during the ED and GD measures and not the females during the GI measures. Since deer are polygamous, the number of offspring does not significantly depend on the number of males. If the males and females are separately captured in this way, the number of harvests does not vary so much. However, this method is possible because the male horned deer is attractive to a hunter who is aiming for a big game. It will not be possible in the fishing of the mackerel. Hokkaido Government requested the Ministry of the Environment to change the notification, and since 1999 relaxed the catching limit of one deer per day, two females are allowed per day. The hunters are well aware of the significance and goals of the management plan, and they are cooperating to catch a large number of female deer.

Relying on game hunting means using deer as a resource. Currently, various efforts are being made to reestablish the market for deer meat. The Hokkaido conservation and management plan also states that deer sika should be regarded as a common property of the people of Hokkaido.

The Wildlife Protection and Hunting Law (first enforced in 1918) was revised in 1999. There are two major revisions. The first is to transfer the permission rights such as the control of pest animals from the Environment Agency to prefectures in order to be consistent with the Decentralization Act. The other is that deer and serow (*Capricornis crispus*), which have excessively increased, are controlled not by the pest control system but by “Specified Wildlife Conservation Management Plan” that is implemented by a prefectural government. At the same time, the Law, which had been for hunters, should include the purpose of protecting biodiversity. The deer management plan in Hokkaido was recognized as a pioneer of the “Specified Wildlife Conservation Management Plan” of the revised law.

In 2014, the law was again amended into the “Wildlife Conservation and Management Law.” At that time, the wildlife management system was introduced by dividing it into three categories: a “protection plan,” a “management plan,” and a “rare wild animal management plan.” Animals with a rare population to be protected and those to be managed in excess were separated in the first and second categories. In addition, even for endangered species, the Ministry of the Environment (Japan) will directly manage animals that require measures against damage caused by wild animals. Also, not only hunters who have been engaged in game hunting, but also wildlife management experts who have acquired population management technology are allowed to manage populations, such as allowing 1 h after sunset.

10.3 Population Estimation by a State Space Model

As mentioned above, Hokkaido estimated the population size of deer in East Hokkaido to be 120,000. Since the discovery rate by areal survey is not 100%, the true population size is not known by this method. IWC (International Whaling Commission) is also trying to estimate the discovery rate by preparing multiple independent observers. For example, suppose two observers A and B. If the numbers of wild animals found by both A and B, found only by A, and found only by B are a, b, and c, respectively, the population size, discover rates of A and B, respectively denoted by n, p, and q, are given by $n = a + b + c + bc/a$, $p = a/(a + c)$, and $q = a/(a + b)$. We can estimate the number of individuals with more detailed data.

However, a more plausible estimate can be made by combining the estimated population number based on an independent population survey each year and the population dynamics model as follows.

$$N_{t+1} = \exp[r + \xi_t - kN_t]N_t - C_{1,t} - C_{2,t} \tag{10.1}$$

Here, t is the number of years from the initial year ($t = 0$); N_t and $C_{1,\,t}$ and $C_{2,\,t}$ are non-negative variables which mean the population size, the number of culled deer, the number of hunted deer in year t, respectively; and r and k are non-negative, which mean the intrinsic rate of population increase and the magnitude of density effect, respectively; $\boldsymbol{\xi_t}$ is a normal random variable with mean of 0 and SD of σ_r, which represents the annual fluctuation (process error) of r. The number of catches in year t is set as $C_t = C_{1,\,t} + C_{2,\,t}$. In this way, the dynamic model that describes the time change of the state is called the "state model" (Yamamura et al. 2008).

We assume that the estimated population (population index) $\widehat{I}_t$ based on population survey and the relative value of the true population N_t have the following relationship:

$$\hat{I}_t = \beta(N_t/N_0)\exp[\zeta_t] \tag{10.2}$$

where β is a positive constant and bias in population estimation, N_0 is the initial population, $\boldsymbol{\zeta_t}$ is a normal random variable with mean 0 and SD is $\boldsymbol{\sigma_N}$, which represents the estimation error of the number of individuals. This is called an observation model. The set of state model and observation model is called "state space model." When β is unknown, only relative values $I_t = N_t/N_0$ will be used in the end, so here the observed value is $\widehat{\mathbf{I}} = \left(\widehat{I}_1, \widehat{I}_2, \ldots, \widehat{I}_T\right)$. In the case of deer management in Hokkaido, the observed value I_t is estimated from the spotlight survey using the generalized linear mixed model (GLMM). From this, the state space model estimate $\hat{I}_t = \widetilde{N}_t/\widetilde{\mathrm{N}}_0$.

From the annual population index estimate $\widehat{\mathbf{I}}$ and the number C_t captured, we estimate $\widetilde{N}_t$, r, σ_r, σ_N, β. To estimate β, we first assume that each parameter value follows a certain probability distribution. This is called prior distribution. For example, assume that r is a beta distribution between 0 and 34%, and σ_r is a lognormal distribution with an appropriate geometric mean and standard deviation (Yamamura et al. 2008).

In the case of sika deer, they breed from the age of 2 years and give birth each year, and the sex ratio at birth is 1:1. Therefore, the number of females at the age of 1 is $N_{c,\ t}/2$, and the number of individuals over the age of 2 is $N_{c,\ t}/2$. Putting the number $N_{f,\ t}$, the number of individuals in the next year is

$$\begin{pmatrix} N_{c,t+1} \\ N_{f,t+1} \end{pmatrix} = \begin{pmatrix} 0 & m \\ S_c/2 & S_f \end{pmatrix} \begin{pmatrix} N_{c,t} \\ N_{f,t} \end{pmatrix} \tag{10.3}$$

Where m is the product of reproductive rate and survival rate from birth to 1 year of age, and S_c and S_f are the annual survival rates of 1-year-old calf and female adult, respectively. When $r = S_c = S_f = 1$, the maximum eigenvalue of this right-hand side matrix is $\left(1+\sqrt{3}\right)/2 \approx 36\%$, which is the mathematical upper limit of the natural rate of population increase. A higher natural increase rate than this limit is sometimes estimated, but it is probably caused by immigration from the outside or an observation error (Matsuda et al. 1999).

According to the probability distribution, the initial population $\widetilde{N}_0$ and the parameter value are given to repeat the computer simulation, and the time series of population fluctuation $\{N_t\}$ and population index are calculated. We generate the sequence $I_t = N_t/N_0$. Calculate the likelihood that an observation will be observed according to the observation model.

$$\Pr[\widetilde{N}_t|N_t] = \frac{1}{\sigma_N\sqrt{2\pi}}\exp\left[-\frac{(\widetilde{N}_t - N_t)^2}{2\sigma_N^2}\right] \tag{10.4}$$

We calculate the posterior probability distribution for each parameter, using the total log-likelihood in the following, denoted by L (Yamamura et al. 2008):

$$L = \sum_{t=0}^{T-1}\log\left[\Pr\left[\widetilde{N}_t|N_t\right]\right] = -\frac{T}{2}\log\left[2\pi\sigma_N^2\right] - \sum_{t=0}^{T-1}\frac{\left(\widetilde{N}_t - N_t\right)^2}{2\sigma_N^2} \tag{10.5}$$

In the Bayesian estimation method, each of the above parameters is given as a prior probability distribution, and the posterior distribution is calculated. That is, when the prior distribution of a certain parameter x is $\Pr[x]$, a numerical experiment of population dynamics model is repeated and numerical experiments on population dynamics model are conducted to find the probability that the observed values become $\widetilde{N}_t$ or $\widehat{I}_t$ when the number of individuals is $\{N_t\}$. Therefore the posterior probability distribution is given by

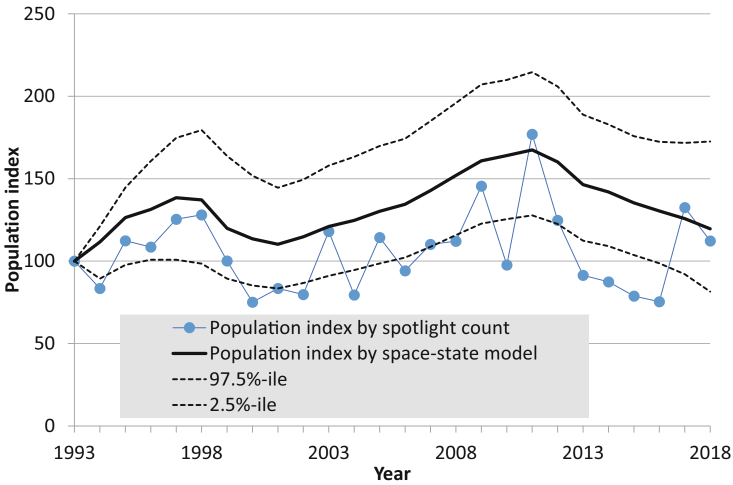

Fig. 10.4 The GLMM population index $\hat{\mathbf{I}}$ (blue circles) using spotlight survey data and the eastern deer population index $\tilde{\mathbf{I}}$ using the state space model. Annual changes in median (solid line) and 95% credible interval (upper and lower dashed lines) (redraw from Data by Hokkaido government)

$$\Pr[N_0 \mid \widetilde{N}_t] = \frac{\Pr[\widetilde{N}_t \mid N_0]\Pr[N_0]}{\int \Pr[\widetilde{N}_t \mid x]\Pr[x]dx} \tag{10.6}$$

When this integration is performed, a large number of random numbers are subtracted according to the prior distribution to generate parameter values, the population dynamics model gives the probability, and the posterior distribution is calculated therefrom (Yamamura et al. 2008) (Fig. 10.4).

Although we agreed to catch a large number of deer, we cannot say that population control had been successful. Although it seems to have decreased for 3 years since 1998, it was far behind the goal of leading to an index of 50% within these 3 years. After that, very regrettably, the budget did not continue and the catch in number decreased. We continued emergent reduction measures, but the target number of catches recommended by scientists (Hokkaido Sika Deer Management Plan Experts Council, formerly the Sika Deer Conservation Management Plan Advisory Committee) will not be achieved, and the deer population had increased again. After that, it seems that the management effort increased again and it began to decline again since 2011, but it is still far below the goal of reducing the population index to 50 by 2022.

Population control is a part of the overall management plan, including building fences to protect the agricultural and forestry fields. In Hokkaido, we are also promoting the effective use of deer meat. We have established more than 100 treatment stations based on the Food Sanitation Law, but the utilization rate of captured individuals is about 20%, as of 2019. It is unrealistic to use it 100% because it includes deer caught in the mountains, but in Hokkaido we aim to make effective use of 30–50%.

In any case, wildlife management and fisheries management, in which management theory is practiced, is a field where scientists are very responsive. It can be seen that the management theory of the two has a lot in common (Shea et al. 1998).

10.4 Yaku Deer is Increasing

Yakushima, inscribed as the Natural World Heritage in 1996, is an island with an area of about 500 km^2. Yaku deer (*Cervus nippon yakushimae*) and Yakushima macaque (*Macaca fuscata yakui*) are endemic subspecies. Population sizes of both subspecies are increasing. Especially in the western part of the protected area, forest floor vegetation has been damaged due to the rapid increase of deer. Fig. 10.5 shows the zoning of the biosphere reserve and density distribution of deer in Yakushima Island. Yakushima World Heritage site is almost the same as the terrestrial core area of the BR. It can be seen that the high-density area of deer overlaps with the western side of the World Heritage area and the western national forest area. There is a national forest area in the south, but there are many orchards on the coast, and it may be that the capture pressure was high to prevent damage to agriculture and forestry (Fujimaki et al. 2016).

There is also a report that the population density is ca. 70 deer/km^2 in the western forest road area, which is particularly dense in the western part (Tsujino et al. 2004), whereas the maximum density of the western area is >100/km^2 in 2018. Usually, when the density of deer reaches >10/km^2, natural vegetation is affected and grasses

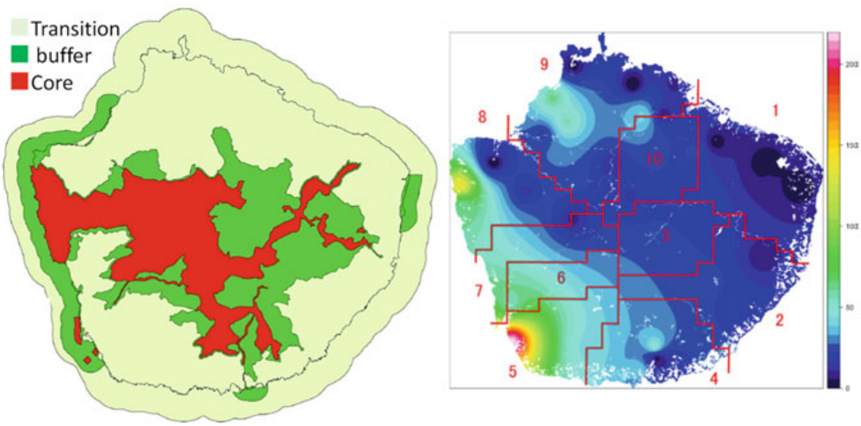

Fig. 10.5 (left) Zoning of Yakushima-Kuchinoerabujima Biosphere Reserve (UNESCO site) and (right) Population density and 10 management units of deer in Yakushima Island, 2018 (Kagoshima Prefecture, Science Council for Yakushima World Heritage, June 25, 2020)

Fig. 10.6 Hananoe-go in Yakushima National Park in 2010 (left) and 2016 (right). The *Sphagnum* moss was peeled off in 2016 due to the trampling of the yaku deer (*Cervus nippon yakushimae*) (Forestry Agency (Japan) 2017 Yakushima World Heritage Science Committee on August 2, 2017)

about 1.5 m tall are cleaned up from the forest floor. This band-shaped space is called a browsing line. Natural vegetation is lost and the ground is exposed in the high-density region as seen in Fig. 10.6(left), but natural vegetation is maintained in the southern part.

Although the number of confirmed species varies among grids, comparing the number of confirmed threatened (sub)species in each of the six areas, the number of threatened species in the western area was small and the numbers in the center and the northeastern were large (Fujimaki et al. 2016). Mt. Miyanoura in Yakushima is the highest peak in the Kyushu region, and it has been registered as a World Natural Heritage site for its diverse biota at various altitudes. The ironic result is that the secluded areas along the coast except the west are more diverse than in World Heritage areas.

The request from the Japanese Society of Plant Systematics (JSPS) in 2003 concerned that many species would become nearly extinct. Most of the surveys by the JSPS are conducted along sidewalks. The topography of Yakushima is extremely steep. Endangered plants could survive in a place where humans and deer cannot access. For example, plants that grow over rocks will avoid feeding damage.

The ecosystem of Yakushima is known to have been devastated by the eruption of a submarine volcano (Kikai Caldera) ca. 40 km northeast ca. 7000 years ago. However, after that, the natural vegetation recovered. However, since we have endured various natural disturbances up to now, there is no guarantee that we will be able to withstand the current human disturbances. At the Shiretoko World Natural Heritage site, a similar controversy was discussed at the Science Council. Although the number of sika deer is still increasing, we have agreed that the impact on natural vegetation such as Japanese elm (*Ulmus davidiana* var. *japonica*) will be the largest ca. 200 years ago, but we could not determine if this was an unprecedented impact in the longer term. There can be both an error of neglecting necessary measures and an error of making unnecessary intervention. These are called type I and II errors in statistics, respectively. In ordinary science, the first priority is to avoid mistakes, or "in dubio pro reo." However, regarding environmental issues, it is recommended that priority should be given to avoiding type II error when irreversible effects are a concern (Matsuda 2003). This is the precautionary principle described in Chap. 3.

A well-known international agreement stating the precautionary principle is the 15th Principle of the Rio Declaration at the 1992 Earth Summit. The UN Framework Convention on Climate Change, which was adopted in 1992, is also based on this precautionary principle. Countermeasures to climate change were taken based on the precautionary principle until the Intergovernmental Panel on Climate Change (IPCC) determined that climate change was a fact in the fourth report of 2007. Even in Shiretoko World Heritage site, it was agreed that deer population is controlled to avoid concerns of irreversible damage to the natural vegetation. We explained this to the UNESCO and International Union for Conservation of Nature (IUCN) investigative bodies in 2008. They understood the measures conditionally. Permits for human intervention, such as deer capture at natural world heritage sites, are probably rare worldwide.

To reduce the deer population of the whole island, the catch in number must be increased remarkably. A short catch will not reduce the population. To test another hypothesis that the effects on natural vegetation are not serious and are not irreversible, we recommend leaving deer growth uncontrolled. Therefore, on the condition that they are caught in the center of the female deer and the number of catches is increased from 300. We concentrate on catch of deer in the northeast and south if the catch increased to 1000. We could control the deer of the whole island if the catch increases to 4000. We proposed the "three-division management plan" (Fujimaki et al. 2016).

In order to reduce the deer population, we need to significantly increase catches, but there are restrictions on where deer can be caught. Before the hunting season in the autumn of 2007, female deer could only be caught by culling. After that game hunting of female deer became possible. In addition, access to hunters for deer catch is restricted within the national forest. This is due to the accidental death of a forest ranger of the Yakushima Forestry Office in 1988.

However, the catch in number has increased significantly since then, and the Yaku Deer Management Plan by Kagoshima Prefecture has been implemented since 2010, and the catch, which was 325 in 2009, has rapidly increased to 4900 in 2012 (over 2000 female deer). If the population of the whole island is probably <20,000 and the rate of natural increase is 20%/yr., we expect that if the whole population will decrease. However, catch in the western part of the world heritage site that extends to the coast and the central part of the mountainous area was difficult. Population control of yaku deer depends on the consensus of local stakeholders.

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Chapter 11
Risk of Avian Collisions in Wind Turbines

Hiroyuki Matsuda, Hiroshi Sugimoto, and Yasuo Shimada

Abstract Utilization of renewable energy is essential for climate change mitigation measures. Wind power generation, solar power generation, geothermal power generation, and biomass power generation have already been put into practical use, and numerical targets of establishment of these stations have been set for reducing carbon dioxide emissions in many countries. However, birds and bats collide with wind turbines. Opposition movements are taking place in various parts of Japan, including wild bird conservation groups. The raptor white-tailed eagle (*Haliaeetus albicilla*) actually has many collisions in Japan. This species has a resident population and a wintering population in Japan and is listed as an endangered species in Japan. Endangered white-fronted geese (*Anser albifrons*) and swans, whose populations were once reduced due to endocrine disrupters including DDTs, are also strictly protected, which are used to be a reason for reductions in the number of installations and operational restrictions of wind turbines, in the environmental impact assessment. However, there are no or very few cases of migratory birds colliding. Introducing the concept of risk into environmental impact assessments cannot reduce conflicts to zero, but it could reduce the number of conflicts so that they do not affect population persistence. For that purpose, adaptive risk management that monitors the number of individuals and the number of collisions and limits the operation of the facility when the number of collisions becomes excessive is effective. In addition, we will explain what kind of consensus was reached on the flight route of white-fronted geese.

Keywords Avian collision · Environmental impact assessment · Joint fact-finding · Adaptive management

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11.1 Future of Wind Power Generation

It has been said for half a century that crude oil will be depleted in the next 40 years. However, the oil age is still continuing partly because of the rise in the price of crude oil. It has become profitable to mine crude oil from undersea oil fields and the discovery of new oil fields. Still, the oil age will be over. “The Stone Age did not end for lack of stone, and the Oil Age will end long before the world runs out of oil.” the quotation is from Sheikh Zaki Yamani, a past Saudi Arabia’s Minister of Petroleum in 1970s, which appeared in The Economist, July 24, 1999 issue.

Fossil fuels are recognized as a major cause of global warming. The 4th Assessment Report of IPCC described “Warming of the climate system is unequivocal” (IPCC 2007). In other words, countermeasures against global warming had been agreed as a precautionary principle without full scientific certainty. Anyway, reducing carbon dioxide emissions has become one of the highest priority environmental problems in the world. Energy that does not emit greenhouse gases includes nuclear power, biomass fuel, solar power, geothermal heat, and wind, which is the subject of this chapter. Geothermal power generation is popular in Iceland, which is an island of volcanoes and hot springs (Fig. 11.1). Biomass fuel absorbs carbon dioxide from the atmosphere during plant growth, but biomass fuel emits CO_2 when burned.

Therefore, the net CO_2 emission itself is almost zero in biomass fuel. However, when cutting a forest to make a maize field, as in the case of bioethanol fuel using maize that has become popular in recent years, a large amount of carbon dioxide is generated during deforestation. Solar power generation does not emit greenhouse gases at all when it generates electricity, but it emits greenhouse gases when

Fig. 11.1 Geothermal power generation is thriving in Iceland (Photo by H.M., June 2006)

manufacturing batteries. This is called indirect GHG emission. According to the life cycle assessment, which includes indirect emissions, solar power generation and other new energies are not completely free of CO_2 emission

Among them, wind power generation is superior to solar power generation in that it emits less greenhouse gases, but inferior to nuclear power and hydropower. Nuclear power has another serious problem such as safety, and it remains a finite resource like fossil fuels. Hydropower has limited locations where it can be developed, and its impact on ecosystems is far greater than wind power.

11.2 Suitable Location for Wind Power Generation

In order to build wind power generation, the annual average wind speed of about 50 m above the ground is at least 21 km/h (ca. 6 m/s), and preferably 25 km/h (ca. 7 m/s) or more. According to the wind condition map created by the Japan Meteorological Association with a resolution of 5 km, it is clear that there are few suitable location conditions from the wind conditions in western Japan and many in Hokkaido. However, in more detail, even in areas where the wind is weak on the wind condition map, there may be local points of high average wind speed.

Furthermore, there are many suitable locations with high average wind speed exist in nature parks. The wind farm is the restriction of construction of wind farm in natural parks such as national parks, quasi-national parks, and prefectural parks is one of the reasons why wind power do not progress in Japan. The areas of quasi-national parks are designated by the Ministry of the Environment, Japan, similar to national parks, but are maintained by the prefectural governments, similar to prefectural parks. The legal framework for all three categories of nature parks is Nature Park Law. Nature parks in Japan occupy 9.1% of the national land (land area), but about 60% of them are estimated to have wind speeds of 21 km/h or more (Nagai and Sera 2005).

Unlike nature parks in the United States, nature parks in Japan include private property areas and areas of residents, forestry areas, farms, and fishing grounds. Nature parks in Japan consist of special protection areas, type I, II, III special areas, and common areas. Land development is regulated in special protection areas and type I special areas, whereas it is possible in other areas if the land owner declares it to park rangers. That is why nature parks in Japan are larger part of land areas than the United States.

Appropriate land for wind power generation is comprehensively judged from the balance of economic efficiency and low environmental impact, especially noise, landscape, and areas where there is no high risk of avian collision.

In addition to the wind conditions just described, areas that satisfy conditions of few noise problems, nearby big power lines, roads for carrying blades and towers of wind turbines, and no land-use regulations. Appropriate land is comprehensively judged by the balance between economic efficiency and environmental impact. Environmental impacts are especially the risk of noise, landscape degradation, and avian collisions. Depending on the wind turbine model, noise problems can occur if the turbine is within 300 m of a residential area.

11.3 Risk of Avian Collisions Due to Human-made Structures

Wind power generation stands in places with sufficiently strong wind speeds, but it may hit the migration routes and habitats of raptors and migratory birds, especially threatened species. As a result, there have been reports of accidental deaths due to collisions with endangered raptors. As a result, there are many planned sites where wild bird conservation groups oppose the construction of wind farms.

Many birds lost their populations in the middle of the twentieth century. One of the reasons is said to be an endocrine disruptor such as Dichloro Diphenyl Trichloroethane (DDT). High levels of contaminants, including persistent organic pollutants (POPs) have been associated with breeding failures and reproductive success in some birds. These substances have endocrine disrupting activity causing thinning of the eggshell and decreased breeding success. The adverse effects of Poly Chlorinated Biphenyl (PCBs) exist in a wide variety of birds (Sakellarides et al. 2006). The adverse effects have sometimes resulted in population decline of several waterbirds (Murata et al. 2003).

There is a global movement against wind farm construction from the viewpoint of wild bird protection. They either oppose the construction itself or request for a plan change. But some environmentalists are not against all wind farms (Bush and Hoagland 2016). For example, the Audubon Society, a conservation organization in the United States, says "Audubon strongly supports properly sited wind power as a renewable energy source that helps reduce the threats posed to birds and people by climate change" (https://www.audubon.org/news/wind-power-and-birds). They also "advocate that wind power facilities should be planned, sited, and operated in ways that minimize harm to birds and other wildlife, and we advocate that wildlife agencies should ensure strong enforcement of the laws that protect birds and other wildlife" (same site as above). The World Wildlife Fund for Nature (WWF) makes a similar claim.

Aside from the importance of countermeasures against global warming, the avian collision risk of wind turbines cannot be said to be higher than that of other artificial structures (Table 11.1). For example, an avian collision of white-tailed eagle (*Haliaeetus albicilla*) flying from Sakhalin to Cape Soya during the snow season may be difficult to detect at Cape Soya in winter, so there may be cases other than those observed. Table 11.1 shows estimates of avian collisions considering the discovery rate.

The impact on the bird population depends on the number of birds in question and the intrinsic rate of natural increase. When a natural monument such as a white-tailed eagle and Steller sea eagle (*Haliaeetus pelagicus*) may collide with a wind turbine,

Table 11.1 Accidental mortality of birds due to artificial constructions (modified from Shimada 2008)

Item	Number of dead individuals	Remarks
Automobiles	60–80 million	Four million miles of total road length
Buildings/windows	98–980 million	4.5 million buildings and 93.5 million houses
Power transmission line	Tens of thousands—174 million	Total transmission line 500,000 miles
Communication tower	4–50 million birds	80,000 towers
Wind power generation	10,000–40,000	15,000 facilities

the Agency for Cultural Affairs, Japan, is concerned with avian collision itself rather than worrying about its population sustainability. Although collision risk cannot be reduced to zero at all, it is possible to reduce the risk of collision. It is one of the main challenges of ecological risk management.

11.4 Wind Power Avian Collision Risk Assessment

To predict in advance how many birds will collide the wind turbine, Sugimoto and Matsuda (2011) used a risk model that assesses the species and number of individuals passing close to the wind turbine. The avian collision risk depends on the cross-section of the sphere with the blade length of the wind turbine as the radius to the planned wind turbines. Bird observers count the number of birds and their trajectories from a fixed point. For the planned site and its surroundings, the observers recorded the size of the field of view that the observer can see from a fixed point, the altitude, and the observation date and time. Based on these data, the annual number of passing wind turbine is estimated.

After the wind farm has been constructed, we expect that the bird avoids the wind turbines to some extent. At Cape Sada on Shikoku Island, Japan, it has been recorded that the flight routes of migratory birds changed after the construction of wind turbines. It has been reported that when a wind turbine was built at the feeding ground for gun ducks, the feeding ground near the wind turbines was no longer used. Losing access to feeding grounds may have a negative impact on fertility and carrying capacity, even if it does not lead to avian collisions. If the migration route changes, there may be negative effects such as extra effort being spent on the migration. However, it is unlikely that the loss of energy, for example, due to a bird flying an extra 1 km, will have a significant impact on fertility and survival.

If birds can detect wind turbines during the daytime, we expect that the bird avoids turbines. Depending on the model of the turbine, a three-blade 2000 kW wind turbine will make one revolution in 2.5–4 s. Flying bird may possibly slip through

Fig. 11.2 Altamont Wind Farm in the United States (photo by Tetsuya Akita, August 4, 2007)

the wind turbine. The grasslands and wastelands where the wind turbines are located are feeding grounds for raptors (Fig. 11.2). When raptors are concentrated on their prey, they are not attentive to the turbines and are at increased risk of collision and death.

At an offshore wind farm in Denmark, radar to identify the flight routes of migratory birds was used (Desholm and Kahlert 2005). Although the altitude of flying birds is unknown, the number of flocks of birds passing through the facility before and after the installation of the wind turbine decreased from 40.4% to 4.5%. The number of birds that entered within a radius of 50 m from the wind turbine tower after the installation was 0.6% of all birds that were detected (Desholm and Kahlert 2005). From this, it is estimated that the probability of avoiding wind farm area is 89%, and even if they enter the wind farm, the avoidance ratio of birds is 92%. Also, if birds jump into the sphere that can be reached by the blades of a wind turbine, the probability of collision is estimated to be about 11%. It takes 3 s for each of the three blades to make one rotation, there is a high probability that they will slip through the blades even if birds jump in the sphere (Sugimoto and Matsuda 2011) (Fig. 11.3).

The planned site for a wind farm in Awara City is located between Lake Katano Kamoike, which is a Ramsar Convention registered site, and the Sakai Plain, which is a feeding ground of white-fronted geese (*Anser albifrons*) and bean geese (*Anser fabalis*). Twice in the morning and evening almost every day, more than 2000 geese move between Katanomomoike and feeding grounds during the wintering season.

Fig. 11.3 Flock of white-fronted geese (*Anser albifrons*) flying over Lake Kitagata. From the Awara wind power plant planned site. Together with the rural landscape in the background, it has a magnificent view. We can see this scene at a fixed time in the sun-rise and sun-set time during the wintering season of geese (Photo by Hiroshi Sugimoto)

Table 11.2 Comparison of avian collision risk between EIA by the business company and the Wild Bird Society of Japan at Awara Wind Farm (Sugimoto and Matsuda 2011)

	EIA by the business company	Wild Bird Society of Japan
Number of surveys (n)	103	39
Observed number of flocks arriving at the planned site (m)	2	4
Average flock size (ν)	200	1632
Number of passages per day $(m\nu)/n$	3.9	167.4
Predicted number of collisions with avoidance	0–2	44,195
Predicted number of collisions without avoidance	42–71	1700

However, most of them fly over Lake Kitagata to the southeast of the planned site and further southeast, but rarely northwest of the planned site.

The collision probability, denoted by P, per bird per single path between the nest and feeding ground is the product of the probability of the following three probabilities, P_1 is the probability of passing through the planned site; P_2 is the probability of flying at the height of the wind turbine; and P_3 is the probability of passing within the area of the rotor disk.

Even if the method is the same, the conclusion changes depending on the result of the data. Table 11.2 shows the results of the preliminary assessment of avian collision risk at Awara Kitagata Wind Power Plant. In the assessment before building the wind turbine, the business company conducted a survey 103 times, most of which the white-fronted geese passed over Lake Kitagata, but only twice passed through the planned site. In the same year, a survey conducted by the Wild Bird Society of Japan resulted in 4 out of 39 trips. The difference in frequency of passing over the planned site between these two surveys is statistically significant. If the geese fly between Katano Kamoike and Sakai plains about 360 times a year, the number of trips to the planned site will be about six times according to the survey results by the business company. On the other hand, according to the survey results of the Wild Bird Society of Japan, it is expected that the number of trips to the planned site will be about 30 times. Moreover, the collision risk is very different because the big flocks were flying (Table 11.2).

However, if the results of both parties surveying the same place are listed together, an additional survey can be performed to verify which is more accurate after the fact. First, it is important to recognize common facts. Joint fact-finding is a process in which two opposing parties confirm the facts together (Matsuura and Schenk 2016).

Regarding the altitude, the probability P_2 of flying at the height of the wind turbine (30–110 m above the ground) is 105 out of 133 (79%), the cross-sectional area of the planned site is 80,000m^2, and the cross-sectional area of one wind turbine is 5027m^2. There are 5 turbines in 2 rows, 10 in total.

Considering avoiding wind turbines in the Danish case above, the probability 1-P_3 of not colliding any wind turbine is estimated to be [1-(1–0.922) $\times$ 0.13 $\times$ 5027/40000]^5 $=$ 99.3%. Therefore, $p = P_1P_2P_3$ is 0.0081% when the wind turbine is not avoided and 0.000072% when it is avoided (Sugimoto and Matsuda 2011).

Geese fly in a formation, but when they encounter a wind turbine, the formation may be disturbed. Assuming that each individual independently collides with the above probability, the probability that a single bird will not collide about 360 times in the morning and evening for half a year is (1-P)^{360}, which is 16% and 2.6% in cases of avoiding and not-avoiding, respectively.

From this, it is predicted that 87 and 0.78 of the 3000 geese may collide if geese avoid and do not avoid the turbines during the winter. If a goose avoids turbines, as much as observed in the Danish case, the number of collisions is sufficiently small. On the other hand, the annual mortality rate is 2.6% if a goose does not avoid the turbine. Although this mortality is not so high to be extinct, the goose is a natural monument and this mortality may be unacceptable by the society.

11.5 How Many Collisions are Allowed?

The upper limit of the number of collisions affecting the population can be obtained by applying the PBR (Potential Biological Removal) of marine animals introduced in Chap. 9. PBR has also been applied to the upper limit of human deaths in threatened birds (Sugimoto and Matsuda 2011; Watts 2010). Katanokamoike Pond has 3000 geese and the intrinsic rate of natural increase is unknown, but it is estimated to be 12%. However, if the safety factor is set to 0.5 considering that it is a special natural monument by Agency for Cultural Affairs, the PBR for this population will be $3000 \times 0.12 \times 0.5 \times 0.5 = 90$ birds per year. Since there is no known human-caused mortality rate other than collision death by a wind farm, this can be regarded as the upper limit of the number of collision deaths. Keeping the number of collision deaths lower than this means providing more protection than the endangered marine mammals. Few voices are concerned about the effect on the population degradation.

The actual number of collisions depends on the validity of these assumptions, and we do not know until the wind farm is built. As with other risk management, the accuracy of prior forecasts is limited. The above calculations assume that the wind turbines are always facing the bird. In addition, the abovementioned frequency of passing over the sky includes the case where birds pass through a corner of the planned site. Therefore, the risk of collision are probably overestimated (Sugimoto and Matsuda 2011).

Therefore, it is desirable to prepare in advance maintenance measures to reduce the risk of collision for the case when a collision occurs more than expected. In the Awara Wind Farm, we consider monitoring the morning and evening arrivals of the geese and stopping the wind turbines when passing over the wind farm, if several collision deaths happen. Since the wind turbine can be stopped in about several seconds, we expect that turbines can be stopped even after an observer detects geese approaching some turbine. Assuming that the facility at 20,000 kW is stopped for 30 min and the power selling price is 10 yen/kW, even if the maximum output is assumed, there will be a loss of about 100,000 yen. It depends on the annual capacity factor (usually about 30%), but it is acceptable to stop the wind turbine about 10 times a year (Sugimoto and Matsuda 2011).

In this way, there are many cases where we use unverified assumptions in risk assessment. There are two ways to make up for it. First, there are cases where safety factors are taken into consideration. For example, when setting environmental standards for chemical substance concentrations and the daily tolerable intake (TDI) of foods, concentrations that are one-tenth lower than those obtained by risk assessment may be set as a safety standard. At this time, it is said to "expect the uncertainty factor of 10 times." However, when using a number of unverified assumptions, we may expect an uncertainty factor of 100 or 1000 times. In some case, it may be an unrealistically strict standard.

Another method that is effective in such cases is adaptive management. One of them is the measures against avian collision of wind power generation planned in Awara City. The risk of a bird's collision is often unknown until it is actually constructed, but we can look around after the fact and check whether a collision death has actually occurred. In facilities with more than 10 wind turbines, there are turbines that a bird is likely to collide depending on the location of turbine. For example, in the wind power facility group of Altamont in the United States, where there are more than 1000 wind turbines, there are cases where the wind turbines at the ends are easy to hit, and the blades are removed leaving only the tower.

Accumulation of collision death findings will enable more accurate risk assessment (Sugimoto and Matsuda 2011).

The Awara wind farm is located in farmland, and it is expected that the detection rate of collision death will be high as it will be visible to residents. Most of the collision deaths are likely to be concentrated around 30 min each during the morning and evening movements of the geese between roosting and feeding sites. The impact of birds changing their travel routes is probably not so large because most of their daily travel routes are not above the facilities but above Lake Kitagata. If we know the wind direction and weather that make it easy to fly over the facility, it can be useful for countermeasures. Even if compared to other wind farms, reactive measures of Awara wind farm are easy.

11.6 Adaptive Risk Management Model for Avian Collisions

In the previous section, we explained the risk assessment of avian collision but we have not examined the effect on the population. The original endpoint (assessment endpoint) of avian collision is not the death of a single bird due to collision, but the persistence of the population. If global warming progresses, it is possible that the geese will stay in other wintering areas, such as the boundary areas between North and South Korea, and will not go south to Japan. In addition, it is said that the geese flying to the Tohoku region of Japan moves southward from Sakhalin, not Korean Peninsula (Takekawa et al. 2000).

The impact of bird collisions due to wind turbines on the bird population is minimal because other accidental deaths are much more common (Table 11.1). In order to evaluate the effect, consider the following population dynamics model. In order to explain the points in an easy-to-understand manner, it is simplified compared to the age-stractured model used for risk management of white-tailed eagle populations that are actually used in Hokkaido.

The number of white-tailed eagles in year t is $N(t)$. The fluctuation of the population can be expressed by the following formula.

$$N(t+1) = \exp[r(t) - aN(t)]N(t) - S(t) \tag{11.1}$$

where $r(t)$ is the internal natural rate of increase in year t, a is the intensity of density effect, and $S(t)$ is the number of white-tailed eagles colliding wind turbines. Accidents other than collisions are considered as natural fluctuations, and here, only collision deaths that can be controlled by the wind farm operator are considered separately.

Stopping the wind turbine for 24p hours will reduce the collision risk in proportion to 1-p. The number of collision deaths depends on the population and the facility utilization rate of the turbines, given by the following binomial distribution:

$$Pr[S(t)=x]=\binom{N(t)}{x}[sq(p(t))]^x[1-sq(p(t))]^{N(t)-x} \tag{11.2}$$

where s is the average collision rate and $q(p(t))$ is the collision coefficient, which is a function of the "outage" rate $p(t)$. Below, it is assumed that $q(p) =$ Max $[4-3p, 0]$ because the white-tailed eagle collides only during the daytime during the overwintering period, stopping the wind turbine for this period (1/4 of the year) almost eliminates the white-tailed eagle's collision. In the Excel file provided on the website (https://ecorisk.web.fc2.com/Springer2021.html), $S(t)$ is given by the following random variable:

$$S(t)=\text{Binom.inv}[N(t), sq(p(t)), \text{rand}()] \tag{11.3}$$

where Binom.inv.[N,p,α] means the inverse of cumulative binomial distribution with a trial number of N, probability of success at each trial p, and the cumulative probability of α. By using a uniform random variable "rand()" for α, Binom.inv. [N,p, rand()] gives the binomial random variable. The average collision rate s is rarely known, but in the following, it is given as a uniform random number from 0 to 0.1.

The actual population size is not accurately estimated. In addition, we do not know the total number of deaths. Therefore, we assume that the estimated number of individuals $N(t)$ and the number of collision death detections $\hat{S}(t)$ are estimated as follows.

$$\begin{aligned}\widetilde{N}(t) &= \text{LogNomr.inv}[\text{rand}(), \log\beta N(t), \xi_N] \\ \hat{S}(t) &= \text{Binom.inv}[S(t), f, \text{rand}()]\end{aligned} \tag{11.4}$$

where LogNomr. inv[rand(), log$\beta N(t)$, ξ_N,] gives a lognormal random variable with a geometric average of $\beta N(t)$ and SD of ξ_N, β is the bias of population size estimation, f is the detection rate of collision carcasses.

We assume that the capacity factor $p(t)$ is adaptively adjusted as follows. First, when the number of collision detections is larger than PBR (potential biological removal, see Chap. 13 for sea lions), which is given by $\tilde{N}e^rF_r/2$, where we assumed that the conservatively estimated population size $\tilde{N} = 600$; the population growth rate $e^r = 0.12$; and the recovery factor $F_r = 0.5$ because the white-tailed eagle is vulnerable in Japanese Red list; and therefore PBR is 9 individuals, the capacity factor $p(t)$ is reduced to 80% of the previous year (Shimada and Matsuda 2007). This is due to the policy that reduces the utilization rate if the influence of the wind turbine on the white-tailed eagle population exceeds the allowable level. Also, when the population estimate for the previous year is less than 600, the collision coefficient $q(p)$ is reduced to Max(0, 4–3[$<\tilde{N}(t)>$/600]). Here, $<\tilde{N}(t)>$ represents the average value of the estimated number of individuals in the past 3 years. If $<\tilde{N}(t)> \leq$

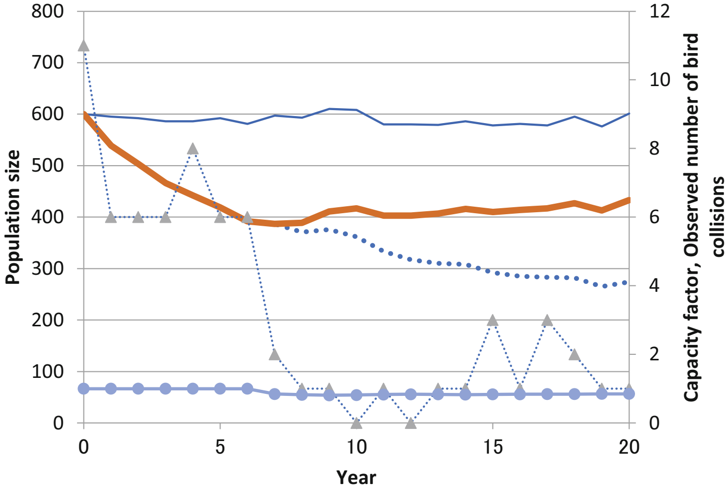

Fig. 11.4 A **simulation result** of population **dynamics** of white-tailed eagle **under** adaptive management of wind turbines. The initial population is 600 individuals, the thin line is the case where there is no collision death due to the wind turbine, the thick line is the case where the capacity factor is adaptively managed, and the dotted line is the case where the capacity factor is always 1. ▲ represents the number of collision detections, and ● represents the capacity factor in adaptive risk management

400, p is reduced to 75% so that collisions will be zero. This is a measure to reduce the impact on the declining population.

An example of the result is shown in Fig. 11.4. If there is a collision death due to a windmill (thick line), the number of individuals will certainly be smaller than if it were not (thin line). However, even if the collision continues, if the utilization rate is adaptively limited, the decrease in the number of individuals can be avoided compared to the case where it is not limited (dotted line), and as a result, the risk of too much population decrease can be reduced. In this example, the capacity factor drops to at least 75%, but if it is lowered to that level, the white-tailed eagle's collision risk can be reduced to almost zero.

When creating an adaptive risk management model, firstly, not only are the fluctuations of the true population size $N(t)$ and the number of collisions $S(t)$, but they are not directly used for management policy. We can use the estimated population size and the observed number of collisions and make a model that changes the policy according to the estimated population $\tilde{N}(t)$ and the observed number of collisions $\hat{S}(t)$. This is the method used in the total allowable catch system in Chap. 7.

Next, we compare cases when (A) no wind turbine, (B) full-time operation of the wind turbine, and (C) adaptively changing operation rate of the wind turbine. By comparing these three cases, it is possible to compare the case where there is no wind turbines risk factor, the case where adaptive risk management is performed, and the case where there is no adaptive risk management. Considering uncertainty, it is

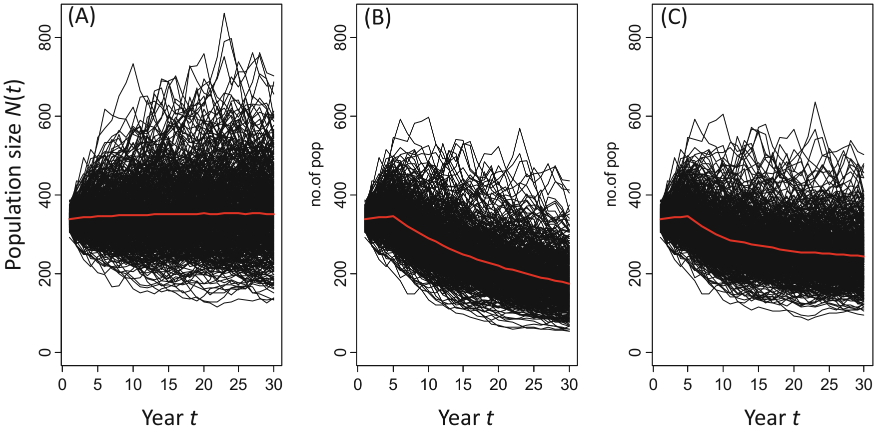

Fig. 11.5 Simulation results of changes in the population size under scenarios when (**a**) there is no wind turbine, (**b**) the wind turbine is fully operated for 20 years, and (**c**) the operation rate is adaptively changed according to the population size. The result of 100 times considering the process error and its average value (red line) (Shimada 2008)

unable to say that a wind turbine has no risk to the white-tailed eagle population. Although it cannot be said, it is shown that the risk can be reduced by adaptive management so that the number of individuals does not decrease below 400 (Shimada and Matsuda 2007) (Fig. 11.5).

If the facility utilization rate decreases, there is a business risk that is less profitable. According to Shimada and Matsuda (2007), the management risk also depends on the power selling price, and at 9 yen/kw, the risk of loss of profit is high, but at 11 yen/kw, the risk is almost zero.

If the power retail price at home is 20 yen/kWh, raising the power selling price allows wind farm companies to take measures to reduce the risk of white-tailed eagle collision without risk of negative net profit. However, if the electricity selling price is lower in Japan, it is more difficult for the wind farm companies to reduce both ecological risk and management risk simultaneously.

In this chapter, we proposed a method for the ecological risk assessment of avian collisions and an adaptive risk management policy as an avoidance measure. At present, wind power businesses depend on the effort of private sectors. At the same time that Japan has been significantly delayed by various induction policies such as the power selling price to regional, electric company. At the same time, it is difficult to take rational measures because it has expressed opposition to the collision of even one conservation group. Risk management models are important in promoting rational consensus building between stakeholders.

The number of wind turbines hit by birds is actually limited. Figure 11.6 shows that from February 2004 to March 2015, 43 cases of white-tailed eagle collision deaths were discovered and reported, 27 of which happened in three wind farms, E, F, and CW. Fourteen incidents happened with two wind turbines at wind farm

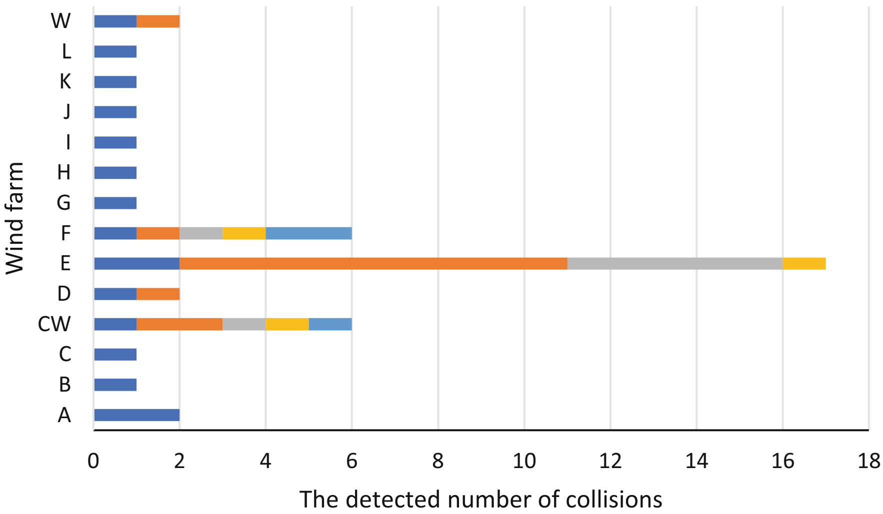

Fig. 11.6 The number of avian collisions detected by anonymous wind power generation facility and wind turbine based on data from the Ministry of the Environment (https://www.env.go.jp/nature/yasei/sg_windplant/birdstrike.html). The different colors on each bar indicate different turbines

E. Only by removing these two turbines, we can reduce approximately 30% of all collision deaths. These turbines stood near the cliffs, and the white-tailed eagle may collide probably when it uses the upwelling. Therefore, refraining from building wind turbines in similar locations will reduce the risk of collision. In fact, about 20 years after their installation, these wind turbines were removed in 2019 and replaced elsewhere.

Compared with Germany and Spain, Japan certainly has many obstacles in promoting wind generation. Since there are many mountains and forests, it is often necessary to cut the forest to create a transportation passage to carry the wind turbine device. There are many typhoons and lightning strikes, which can cause malfunctions. Therefore, the amount of wind power generation in Japan is considerably smaller than other nations like Germany. In the future, there are plans to create offshore wind farms, but there are few shallow beaches suitable for location in Japan. In addition, Japan's coasts have exclusive user rights for fisheries, and it is necessary to coordinate with fishers.

However, there are many coastlines, and floating wind turbines are considered. If fish reefs could be the basis of the offshore wind, it would be easy to obtain agreement with fishers.

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Chapter 12
Resource Economics of Exotic Mongoose Control

Hiroyuki Matsuda, Koji Kotani, and Shigeki Sasaki

Abstract Although it is possible to reduce the number of exotic species by 90%, we often say that we need enormous cost and labor to remove the remaining 10%. On the other hand, if it is not eradicated, it must be controlled forever. In this chapter, we introduce "economics of exotic species control," including mongooses that are introduced into Amami Oshima Island, as an example. Regarding the control of fruit flies, there is a successful example of eradication from small islands by mass release of sterile males instead of capture by traps. The relationship between catch per unit effort (CPUE) and the population size (N) is important, and in many cases CPUE is a concave function of N. In this case, the total effort required for eradication will be infinitely large. CPUE-dependent population estimation involves uncertainty. By combining with a population dynamics model, the nonlinearity of the N-CPUE relationship can be estimated. In order to eradicate with a finite budget, we could expect eradication of the invasive species (1) if its natural population growth rate has the Allee effect, (2) if the population is induced to a sufficiently low density and accidental extinction due to demographic stochasticity, and (3) combining methods such as detection dogs and chemical spraying in addition to capture by traps. The Allee effect cannot be expected so much because a small number of invading individuals have increased and become established. In the case of maintenance at a low population density, it is unknowable when it will be extinct, but the management cost may be cheaper than eradication because the ecological impact is small when the population is small. Eradication may be possible in combination with other active methods. If eradication on Amami Oshima Island, which has an area of about 712 km^2, will be successful in the near future, this practice and its comprehensive analysis, including its cost-effectiveness, is valuable worldwide.

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12.1 Why Do We Need Alien Species Control?

Alien species are listed as one of the five major drivers that impair biodiversity (Secretariat of the Convention on Biological Diversity 2014). Especially in reptiles, alien species are considered the second largest driver after habitat loss. However, on oceanic islands such as New Zealand and Ogasawara, which are defined as an island that has not ever been connected with any continent, we can say that all terrestrial flora and fauna have been imported from outside. The history of alien species is diverse, including invasion in prehistoric age, pre-modernization, post-WWII, and settlements in the twenty-first century. Those that are naturally introduced are not included in the alien species even if they are recent. Some species are introduced consciously by humans or unconsciously imported with humans or cargoes. In addition, even if the same species exists at the destination, they may differ in subspecies or genetically between two locations. Different subspecies are called exotic subspecies, and different genetic strain is recognized as another problem, "genetic pollution" (FAO 2019), especially in the aquaculture industry.

As you may see from the perspective of oceanic islands, even if there is no human influence, organisms move at some frequency to form a newly natural ecosystem. These exotic species may interact with native species and, in some cases, may play an important role in the persistence of native species. It is sometimes argued that only those that have been established more than 100 years ago should be protected as conventional and those that are new should be excluded as exotic, and that they should be judged only at the time of the settlement of invasion. If we do not know whether some species' colonization is due to natural migration or unintentional invasion, opposed measures of protection or eradication may be favored. It is controversial.

Extinction of species at low frequency occurs due to natural phenomena. Extinction of local populations occurs more frequently, and it is sometimes "unnatural" to protect it. Human beings are also a part of the ecosystem, and the nature that excludes human beings is not the ideal nature for human beings. The objective of exotic species control, as with other measures for biodiversity conservation, is to maintain the ecosystem services or "nature's contributions to people" (NCP, Diaz et al. 2018) in a sustainable manner. From the precautionary principle, we like to avoid that the impacts due to invasion of exotic species that are not fully understood but more exotic species have invaded than the natural frequency.

Originally, measures should be taken after elucidating the advantages and disadvantages of exotic species exclusion, but it is difficult. At least as a whole, the adverse effects of exotic species are serious. Therefore, each member state of the

Convention on Biological Diversity is taking measures against exotic species with a certain standard.

The Japanese "Act on the Prevention of Adverse Ecological Impacts Caused by Designated Invasive Alien Species" (hereafter abbreviated by the "Alien Species Act") came into effect in 2006. Under this law, the target of exotic species is limited to species that are imported from abroad. Some parts of "Invasive alian species" (IAS) are designated as "designated alien species" to prevent damage to the ecosystem, human health, agriculture, forestry, and fisheries. The law prohibits feeding individuals, and transplantation and releasing an individual that has been caught, Prohibition, penal regulations are set, and eradication is aimed if appropriate.

The purpose of the "Basic Act on Biodiversity," which came into force as in 2008, is to realize a society in which the gift of nature can be enjoyed by future generations. Although there is not always clear evidence that the invasion of each exotic species or the extinction of native species impairs the gift of nature, it is considered that it probably disrupts native ecosystems or human life from the perspective of the precautionary principle. Non-native species are subjected to extermination and eradication.

As mentioned later, neither the Convention on Biological Diversity (CBD) nor the ecologists consider all alien species to be excluded. The CBD and its member states control exotic species for benefits to people, not "for nature." The nature conservation movement is under evolution. There were many dogmatic debates about the exclusion of exotic species.

Many enthusiasts of game fishers disagree with the eradication of largemouth bass (*Micropterus salmoides*), which is a designated IAS. But it is believed that it will have a great impact on the native species. On the other hand, even if there is a high possibility of species that was introduced by another place within the national boundary, if there is a risk of extinction at the original place of the species, as will be shown later, a destination habitat is subject to conservation depending on the threatenedness of the species.

It was thought that black kokanee (or kunimas, *Oncorhynchus nerka kawamurae*), an endemic species of Lake Tazawa in Akita Prefecture was extinct by 1990. Fortunately, it was rediscovered in Lake Saiko, Yamanashi Prefecture in 2010 (Nakabo et al. 2011). Even if it is an introduced species in Lake Saiko, there may be a way of thinking that it will be actively protected with the value as an ecosystem service.

Whether a species is native or exotic is not an essential issue. We will decide how to treat this species regarding the treatment of the species, along with the harmful due to invasion into native ecosystems, the danger to human health, the impact on agriculture, forestry, and fisheries, the biological history, and the context of local culture after the invasion. Alternatively, it may be important to determine the survival of the entire genetic lineage comprehensively in original habitats and other places after thorough consultation with the relevant stakeholders.

In wildlife management, long-term management is necessary to avoid both outbreaks and extinctions. As for measures against alien species, if they are eradicated successfully, they should be prepared for reinvasion thereafter. If eradication cost is less expensive than permanent control, the eradication option is favored. What is important is not to eradicate the exotic species but to sustainably maintain the ecosystem services obtained from the native ecosystem. Eradication is one of the options.

12.2 Bioeconomics of Measures Against Exotic Species

As mentioned above, alien species are one of the major drivers of biodiversity loss. According to Kotani et al. (2011), many studies on the management of alien species have been published in the fields of ecology and economics. In this chapter, we aim to present theoretical results on adaptive management strategies in exotic species control from a bioeconomic perspective.

This chapter focuses on the feasibility and profitability of the eradication efforts in alien species management (Bomford and O'Brien 1995; Myers et al. 1998). The goal of eradication is easily adopted by regulatory authorities. The advantage is that the options for eradication are clearer than other long-term management goals. Moreover, eradication looks clearly the best solution if we do not take into account the costs. Unfortunately, the existence of such an attractive eradication goal has been reported to be a potential pitfall for regulatory authorities (Simberloff 2002).

Similar to wildlife management, it is relatively easy to reduce the amount of exotic species at least at an early stage. However, if it is not eradicated, the population may increase again. The difficult part of the exotic species control project is that unless it is left standing or eradicated, it must be continued forever.

If native wildlife such as deer and bears that are considered harmful to human society is overabundant, as described in other chapters, the option to eradicate these species is not acceptable in the context of the Convention on Biological Diversity. Therefore, the purpose of wildlife management is both population persistence and damage control. However, it is possible to eradicate exotic species. The eradication is not the goal, but the purpose is to rehabilitate the functioning of the native ecosystem and to reduce the adverse effects on human society. If the number of exotic species is sufficiently reduced, the adverse effects can be reduced. We should recognize that the option of eradication was added to standard wildlife management.

We will examine the adequacy of eradication of exotic species management through bioeconomic modeling. This is a challenge derived from recent evidence that most eradication attempts have failed or were discontinued without success (Myers et al. 1998). We know that the failure of the eradication plan is a tragedy and should not be repeated. A typical case we have observed is that intensive eradication work takes place over a decade, even if eradication is not feasible.

For example, the eradication plan for gypsy moth (*Lymantria dispar*), which started in the late 1900s in North America, has failed (Myers et al. 1998). If the removal process is stopped along the way, the population of alien species will increase again and return to the original level. In this case, the financial costs, resources, and time spent aiming for eradication are completely wasted.

The reasons for the failure of such eradication are now well documented. The literature points out that there are two main factors. The first factor is the density dependence of catchability, which definition is mentioned later. This reflects the fact that catchability may decline as the population decreases (Bomford and O'Brien 1995; Myers et al. 1998; Simberloff 2002). The second factor is the variety of uncertainties associated with the management of alien species (Eisewerth and van Kooten 2002; Olson and Roy 2002; Saphores and Shogren 2005). Applying bioeconomic theory, Kotani et al. (2010) sought the best adaptive management strategy, which provides some important implications.

The components of the bioeconomic model for exotic species are taken from standard bioeconomic models in the classical theory of renewable natural resource management including fisheries (Reed 1979; Walters 1986; Clark 1990). This is because the model can be easily tailored to the management of exotic species in the sense that density-dependent catchability and uncertainty are well understood. More importantly, some theoretical results obtained with such bioeconomic models can be validated with field survey data. First, we introduce a basic bioeconomic model of exotic species management to present results related to density-dependent catchability in relation to optimal eradication decisions. Then incorporate uncertainty into the model and compare the results with those without uncertainty.

12.3 Eradication of Exotic Melon Fly by Mass Release of Sterile Males

A famous example of successful eradication is the melon fly (*Bactrocera cucurbitae*) in Okinawa Prefecture, Japan. This is an agricultural pest that damages cucurbit crops and causes serious damage to various fruits and vegetables (Kakinohana et al. 1997). The invasion of melon fly to Okinawa Islands was confirmed in 1919 on Kohama Island. After that, in 1970, before the Okinawa Prefecture returned from the USA to Japan, it was confirmed that melon fly had invaded into Kumejima Island, so that citrus fruits in Okinawa could not be transported to the inland where no melon fly existed. Therefore, Dr. Yoshiaki Ito and his colleagues conducted a large-scale project to release sterile males of melon fly (Ito 1977). A large number of male fly, which had been sterilized by radiation, was produced in large quantities in a factory and then released outdoors. Although the number of males of that generation increases by the amount they release, the number of males of the next generation decreases because females mated with infertile males cannot reproduce. Agricultural damage is caused by larvae, and the increase in the number of infertile males does not cause agricultural damage. The eradication project that started in 1975 declared eradication on all Okinawa Islands in 1990.

There is a big difference from the extermination of exotic species by traps as described later, the effect of the eradication by releasing sterile males increases as the number of exotic species decreases. We will show it by a mathematical model.

We denote that the number of wild female and male individuals of the exotic species at year t are $N_f(t)$ and $N_m(t)$, respectively; and the number of released sterile males is $N_s(t)$. When there are no sterile males, the population dynamics of an alien species can be expressed as follows.

$$\begin{aligned} N_f(t+1) &= N_f(t) \exp\left[r - a\left(N_f(t) + N_m(t)\right)\right] \\ N_m(t+1) &= N_f(t) \exp\left[r - a\left(N_f(t) + N_m(t)\right)\right] \end{aligned} \tag{12.1}$$

where r and a mean the intrinsic rate of population increase and the magnitude of density effect, respectively. This is called the Ricker equation (Clark 1990). Since males and females are born from females, the first factor on the right side of the second equation is N_f (t). Since the numbers of male and female individuals are always the same, N_f (t) should be substituted for N_m (t) without considering the equation for $N_m(t)$.

We obtain the following equation to incorporate sterile males:

$$N_f(t+1) = N_f(t) \frac{N_m(t)}{N_m(t) + N_s} \exp\left[r - a\left(N_f(t) + N_m(t) + N_s\right)\right] \tag{12.2}$$

Since sterile males are produced in factories, we assume here that N_s is constant each year. Here, we considered the density effect including the infertile males, but when the density in the early stage of life history is effective, the last factor should be $\exp[r - a\ (N_f(t) + N_m(t))]$. In the above Eq. (12.2), the condition that the number of females increases next year is

$$f(N_m) = \left[\frac{N_m}{N_m + N_s}\right] \exp[r - a(2N_m + N_s)] > 1 \tag{12.3}$$

where N_f was replaced with N_m. Using

$$f'(N_m) = \frac{N_s - 2aN_m(N_m + N_s)}{(N_m + N_s)^2} \exp[r - a(2N_m + N_s)], \tag{12.4}$$

The value of $f(N_m)$ is maximized when $N_m = \sqrt{}(N_s^2/4 + N_s/2\mathrm{a}) - N_s/2$. If N_s is sufficiently large, $f(N_m) < 1$ is always satisfied and the eradication is possible (Fig. 12.1). If the number N_s of sterile males released each year is large enough, it is possible to eradicate the exotic species that has been established. Even if the number of sterile males released is smaller than that, it can be eradicated if they are dealt with before the number of alien species increases too much. In the case of Fig. 12.1, if $N_s \geq 447$, it is possible to eradicate regardless of the number of female individuals at the time of introduction of mass releasing of sterile males. Even if N_s is 400 individuals, it can be eradicated if $N_f < 827$.

As shown in Fig. 12.1, the rate of decrease of exotic species increases with decreasing the population of exotic species. Since sterile males are artificially

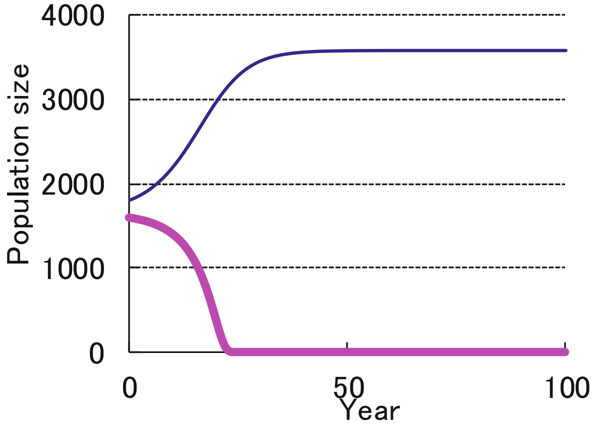

Fig. 12.1 Change in the female population of population dynamics expressed by Eq. (12.2) when sterile males are released into an exotic species. When $r = 0.6$, $a = 0.0001$, $N_s = 400$, if $N_f < 1654$, it can be eradicated (thick line), otherwise the exotic species will increase (thin line)

produced, a certain number of females can be produced regardless of the number of field individuals, and female mating partners have a higher rate of becoming sterile males as the number of wild exotic males decreases.

Long after the eradication was successful, the sterile male production plant is not closed but continued to produce sterile males in preparation for another invasion of melon fly. Even if it invades again, the number of individuals should be small at the beginning of the invasion, so even small-scale release of sterile males can prevent colonization of melon fly.

The biggest concern in controlling by releasing sterile males is that females have the ability to distinguish sterile males from wild males. Although we assumed random mating, it is conceivable to select a mating partner based on some male trait. This is called assortative mating. If the ability to avoid sterile males is provided, it will be very advantageous in terms of selection, so the possibility that such an ability will evolve cannot be ignored.

12.4 Control of Alien Species by Traps

In melon fly, sterile males were released and successfully eradicated, but in many exotic species, capture by traps is the largest eradication measures. Now, the Ministry of the Environment is working on a control project to eradicate the Small Indian Mongoose (*Herpestes auropunctatus*) that was introduced into Amami Oshima Island and Okinawa main island, both of which were nominated as a natural world heritage of “Amami Oshima Island, Tokunoshima Island, Northern part of Okinawa Main Island and Iriomote Islands” in 2017. If a large number is captured, the population will decrease and a certain control effect will be obtained.

However, complete eradication is another matter. We often say that “it takes a lot of effort to capture the remaining 10% of the alien species rather than to capture the

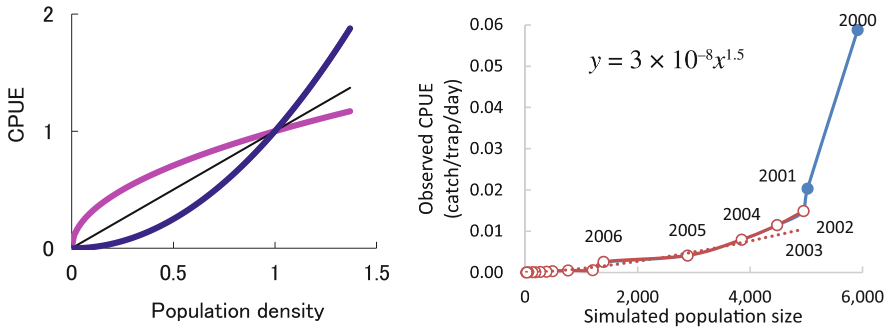

Fig. 12.2 (left) Schematic diagram of the relationship between population density and number of captures per unit effort (CPUE). (right) The relationship between the simulated population size and observed CPUE when $r = 1.4228$, during 2000–2018. The regression curve is obtained from the data during 2002–2018

first 90%." Unlike the case of releasing sterile males, the trapping effect decreases as the population density decreases.

We can describe the following mathematical model:

$$\begin{gathered} N(t+1) = [N(t) - C(t)]f(N(t) - C(t)) \\ C(t) = qN(t)\ \mathrm{E}(t) \end{gathered} \tag{12.5}$$

where $N(t)$ and $C(t)$ mean the population size and the catch-in-number at year t, respectively; $f(N(t))$ means the number of the next generation per individual. Although mongoose breeds in the next year after adults survive, we simply assume that the exotic species that breed and die in 1 year; q and $E(t)$ are the probability of catching an individual per trap per day (which is called catchability in fisheries) and the amount of effort to catch at year t, respectively. Strictly speaking, if a trap is set up for several months, the catchability may be different at the beginning and the end of a certain year. We suppose that the catch-in-number at year t is expressed as $qN(t)$ $E(t)$. In this case, $qN(t)$ or the catch-in-number divided by the effort is called the catch per unit effort (CPUE) and is used as an indicator of population abundance.

If q is a constant, CPUE is proportional to the number of individuals. However, q may depend on the population size, as is assumed by

$$q = q_0 \left(\frac{N}{A} \right)^{\theta} \tag{12.6}$$

where N and A mean the population size and the habitat area, or N/A means the population density per habitat area; θ means the magnitude of nonlinearity in the density-dependent catchability. Although it is often assumed that CPUE is simply proportional to the population density (that is, the catchability is constant regardless of population density, or $\theta = 0$), even in that case, the cost of capturing an individual will increase as the population decreases, unlike releasing sterile males. As shown in Fig. 12.2, when the population size decreases, it may be harder to catch more than it decreased.

When CPUE is convex shown in Fig. 12.2 (left), that is, when θ in Eq. (12.6) is negative, the CPUE is relatively high even when N is close to 0 and we can seek the eradication. When CPUE is proportional to the population density, that is, if θ is 0, CPUE is proportional to the population density. When CPUE is convex, that is, when θ is positive, the CPUE drops sharply when N decreases. When CPUE decreased to 0.3 fold smaller than the present, the population density also decreased to 0.3 if $\theta = 0$. The density decreased to lower when $\theta < 0$. On the other hand, the population density decrease by more than the CPUE decreases when $\theta > 0$.
The sign of θ depends on the condition. If θ in fisheries resources was negative (convex curve in Fig. 12.2), there was a concern that overfishing of fisheries could lead to resource exhaustion. However, if θ in exotic species is positive, the eradication may not be feasible. There is no evidence that the density dependence of catchability is opposite between exotic species and marine resources. It will depend on the assumed density, or q may be an S-shaped function with respect to density (Sasaki and Matsuda 2010).

We will explain three possibilities. (1) θ is positive if the natural resource is spatially unevenly distributed and fishers first operate in the fishing ground with high density and extend their operation area as the stock size decreases. CPUE is considered to be determined by the relationship with the local density. Such an argument was made when the sharp decrease in CPUE of Japanese tuna longline fishing vessels that has been the basis for the "90% reduction theory of tunas" (Myers and Worm 2003). In such a case, CPUE sharply decreases ($\theta > 0$) before the total number of individuals decreases. (2) When the distribution area is shrinking as the population decreases, hunters can catch deer in the center of habitat whose density is high due to aggregation from circumstances, CPUE does not decrease much ($\theta < 0$). For example, instead of fishing boats randomly exploring the sea, if the fishing boats exchange information with each other and the weather information is used to search for high sea surface temperature areas where fish are likely to exist. (3) When there are innocent individuals that are easily caught and smart individuals that are difficult to be captured, the latter remains when the former is captured, so the individuals that are easily captured are more rapidly than the total population decreases (Yoneyama et al. 1992). In this case, θ becomes positive. In Mongoose, etc., individuals that are difficult to catch are called "trap-shy" (Clout and Williams 2009).

The situation of (2) may happen frequently when overfishing continues, but the situation of (3) may occur just before eradication. It can be said that the management of the fishery fails because the resources are not recovered for a long time.

12.5 Eradication Costs

Since 2005, the "Project for the Promotion of the Control of Specified Invasive Alien Species", which includes the eradication project of the small Indian mongoose control project on Ammon Oshima Island, has been spending 300 million yen every year since 2005. Part-time staffs called "Amami Mongoose Busters" were hired to trap a large number of mongoose. Figure 12.3 shows the annual changes in the number of captures and CPUE. In this way, you can see that CPUE is drastically reduced. However, considering that mongoose is expected to have a natural increase rate of about 42% per year, and 40 years have passed since the introduction of 30 mongooses on Amami Oshima Island in 1979. We show the estimation of population trend in Fig. 12.3 by using.

$$N(t+1) = [N(t) - C(t)]e^r \quad (12.7)$$

where we assume that $N(1979) = 30$ and use C_t as the actual catch record. We obtained simulated population dynamics if we assumed $e^r = 1.42282$, 1.422825, and 1.42283. If r is higher than this, the number of individuals should continue to increase. If r is lower than this, it is already extinct. The interpretation is that rather than knowing r exactly from past time series, density effects must exist and/or r varies over time.

Figure 12.4 shows the relationship between the population size calculated from the population dynamics model and the observed CPUE from 2000 to 2018. Although the population dynamics model is not necessarily quantitatively valid, it is certain that θ is positive and the relationship is concave. Therefore, CPUE has decreased to about 3/10,000 within the 14 years from 2000, while the calculated population size did not decrease by the same ratio. However, when the regression

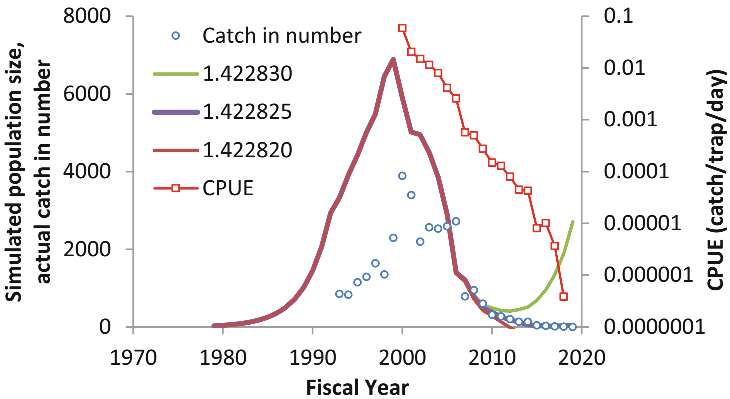

Fig. 12.3 Temporal change in the number of catches (circles) and CPUE (squares) in the mongoose control project in Amami Oshima Island, and a simulated population size obtained by Eq. (12.5) for $1.42282 < r < 1.42283$

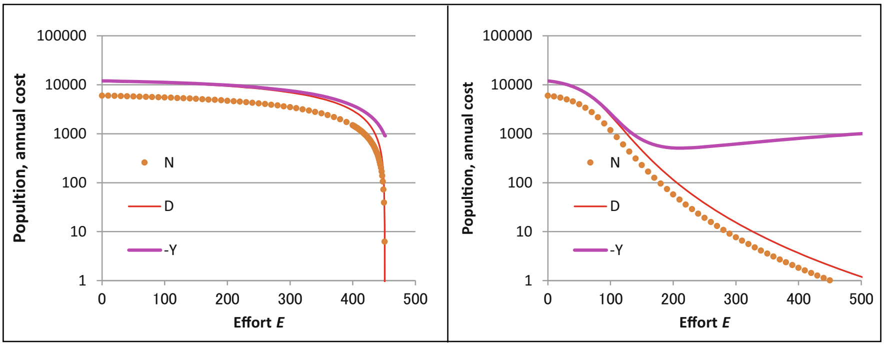

Fig. 12.4 Relationship between capture effort, equilibrium population size (circles), damage (thin line), and total cost (thick line). When the density dependence of catchability is 0 (left) and 0.2 (right). The total cost is minimized when $E = 450$ (left) and $E = 210$ (right)

curve after 2002 is taken, θ is estimated to be ca. 0.5. It is certain that it has drastically decreased and is either close to eradication or population extinction.

In this way, the density dependence of catchability depends on the target species, capture measures, heterogeneity in spatial distribution, and the target density. Furthermore, θ is not constant and may change with time, the distribution area, and effort.

Eradication projects are expensive. It is impossible to pay the infinite cost. If the cost is limited and high, it will be necessary to decide the way of the control business in terms of cost-effectiveness. Regarding fishery resources, we have derived the optimal fishing policy from the capital investment of fishing vessels, running costs of fishing, and fish prices. Regarding exotic species, it is costly to control, and if exotic species continue to exist, the loss of ecosystem services such as the reduction of native species and disturbance of ecosystems may occur. However, it should be possible to find the optimum control policy from the viewpoint of resource economics, only replacing the profit in fisheries by the damage due to exotic species. The population dynamic model (12.5) is extended by considering the process error in the population variation of exotic species. Then, the existence of the exotic species and the total cost for the control (the "yield" is denoted by Y with a negative sign) is expressed as follows, and we can derive the optimal control policy:

$$\begin{aligned} N(t+1) &= [N(t) - C(t)]e^{r - a[N(t) - C(t)]} + \sigma_d \xi(t)\sqrt{[\mathrm{N(t)} - \mathrm{C(t)}]} \\ C(t) &= qE(t)N(t)^{\theta+1} \\ Y(t) &= -bE(t) - D[N(t) - C(t)] \\ Y^* &= \sum e^{-\delta t} Y(t) \\ D[N - C] &= u(N - C)^v \end{aligned} \tag{12.8}$$

where r and a are the intrinsic rate of population increase and the magnitude of the density effect, respectively; σ_d and $\xi(t)$ are the magnitude of demographic stochasticity and the random variables representing demographic stochasticity, respectively; b and $D(N)$ are the cost of eradication per unit effort and the loss on ecosystem services when N individuals of the exotic species exist, and Y^* and δ are the sum of three present values of the cumulative discount cost and the economic discount rate, respectively; u and v are positive constant and the magnitude of nonlinearity for the damage. We use the function form $f(x) = e^{r - ax}$ of the Ricker equation for the reproduction curve $f(N - C)$ in Eq. (12.7).

For simplicity, only the demographic stochasticity is considered here as the process error, and we ignore the environmental stochasticity such as the annual variation of r. Unlike fishery and forestry, it is said that the existence of exotic species has a negative economic value. However, not only the value of natural resource products C but also the value of ecosystem services that exist as forests have been considered for forest resources (Satake and Rudel 2007) and fisheries resources (Matsuda et al. 2010).

When the demographic stochasticity is negligible, whether eradication is possible depends on the density dependence θ in catchability. When θ is negative, the relationship between the capture effort, the equilibrium population size, and the cumulative discounted cost is as shown in Fig. 12.4. The equilibrium population size decreases with increasing the control effort. When θ is positive, the control cost is so high with increasing the cumulative discounted cost that the eradication project is not reasonable. The loss of ecosystem services decreases as the number of individuals decreases. In particular, it is very expensive to capture the last few individuals. If θ is negative and the absolute value is large enough, it is reasonable to increase the amount of effort to eradicate it (Kotani et al. 2011).

Unfortunately, θ is often positive which means the eradication is very difficult, and catchability is likely to decrease as eradication approaches. In order to maintain a constant catchability even when the population density is low, efforts should be made at the already extinct areas rather than the center of the distribution. It is necessary to devise a trap and use of detection dogs that detect exotic mongoose.

We have mentioned some theoretical grounds that make it difficult to eradicate, but there are some possibilities for resolution. First, there are several factors that make θ negative as mentioned earlier. The other is that the reproduction curve $f(N)$ is convex downward ($f''(N) > 0$) when the number of individuals is low and continues to decrease even without eradication effort ($f(N) < 1$). This inverse density dependence is called the Allee effect. It is not clear how much the Allee effect exists, but if the exotic species increased from a low population density when they were

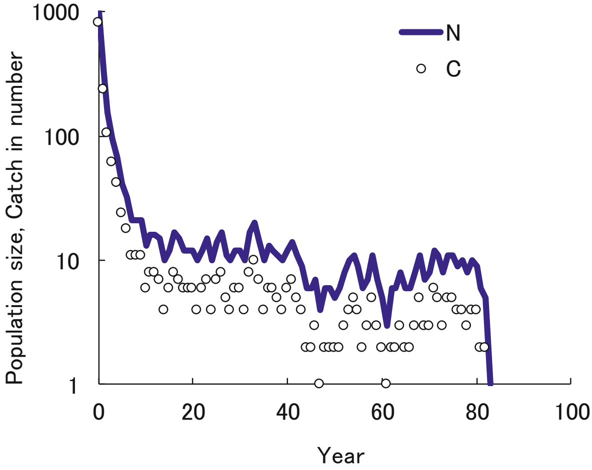

Fig. 12.5 Results of theoretical model of exotic species control effect considering demographic stochasticity. If the population size (thick line) is maintained at 20 or less, the chance of extinction will occur at random due to demographic stochasticity. The circles indicate the number of captures

introduced, the Allee effect would not be a common factor for exotic species that have been successfully colonized.

Another possibility is the effect of demographic stochasticity. Even if it is difficult to keep the number of individuals sufficiently low and reduce it below that, the number of individuals may decrease naturally due to demographic probability. If the number of individuals is 50 or more, the extinction probability due to demographic stochasticity is extremely low, but if it is less than 50, it would not be negligible. Figure 12.5 shows the extinction of an imaginary exotic population when it is maintained at 12 individuals in the deterministic model due to demographic stochasticity. However, it is not known the time when it will become extinct, and it is necessary to keep exotic species under low density for a few to a hundred generations.

Considering environmental stochasticity, it is not optimal to make the capture effort E constant every year. It is best to keep the post-capture population constant as in the theory of fisheries management (Kotani et al. 2009). In other words, the capture rate changes with the population size. Therefore, it is best to adjust the effort every year.

The catchability depends on the population density, but the demographic stochasticity depends on the total population of the habitat. If the total population can be suppressed to less than a few tens when the population density is reduced in a relatively small area, it may be profitable to eradicate it by demographic stochasticity. However, in that case, when it becomes extinct depends on chance, and until it is eradicated, it is necessary to spend a budget and continue strong capture efforts. Without knowing how long the eradication business will last, it may be difficult for the administration to convince taxpayers to keep their budget.

Eradication of exotic species is often costly and labor-intensive. However, eradication is an irreversible event, so once eradicated, it will be easier. However, while the cost of continuing population control can be realized from experience, the cost of eradication may be much higher than expected.

12.6 "Final Stage" of Amami Mongoose Eradication Project

The spatial distribution of the exotic species will extend from the position where it was first released. Figure 12.6 shows the secular change in CPUE of mongoose on Amami Oshima Island. Initially, 30 mongooses were released in Amami City (former Naze City) in 1979, and the population distribution spread to almost the entire island. Although the distribution area is still wide, the density has decreased since the start of the eradication program.

In the case of the mongoose eradication project in Amami Oshima Island, detection dogs have been introduced since 2008, and traps are focused on areas where mongooses still exist. This may improve the catchability. The detection dog itself has also captured mongoose. The number of catches in 2014 was 39 by traps and 32 by detection dogs and their handlers, suggesting that detection dogs can be efficiently captured even if the number of mongoose decreases (Fig. 12.7).

In 2017, the number of mongooses was captured in the Mineyama and Nakayama areas of Yamato-son, and the number of captures in 2017 was only Naze Kominato, Amami City. It is thought that the population was fragmented. There was no capture of mongoose by detection dogs in 2018. A chemical control test was carried out at Mineyama area of Yamato-son, where the remaining mongooses were observed, and it was confirmed that more than one mongoose were eating food containing chemical substances. Sensor cameras were installed in the first half of 2019, but the captured image was not seen. Finally, in 2018 and 2019 respectively, the number of mongoose caught was 1 and 0, respectively. Japan's Ministry of the Environment is

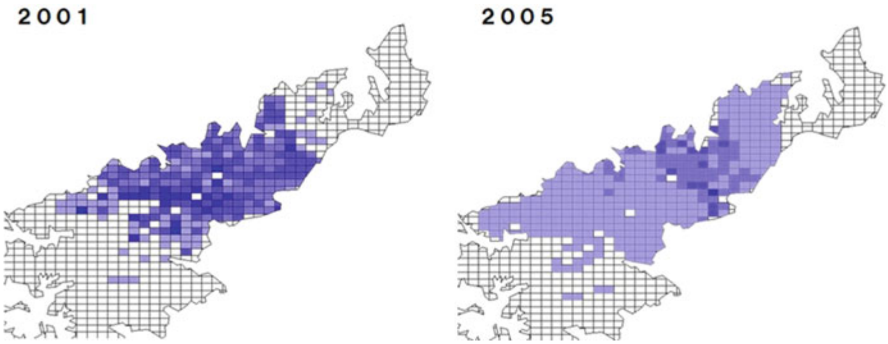

Fig. 12.6 CPUE (captures per 100 traps per day) of the mongooses in Amami Oshima Island during 2001–2005, provided by Japan's Ministry of the Environment

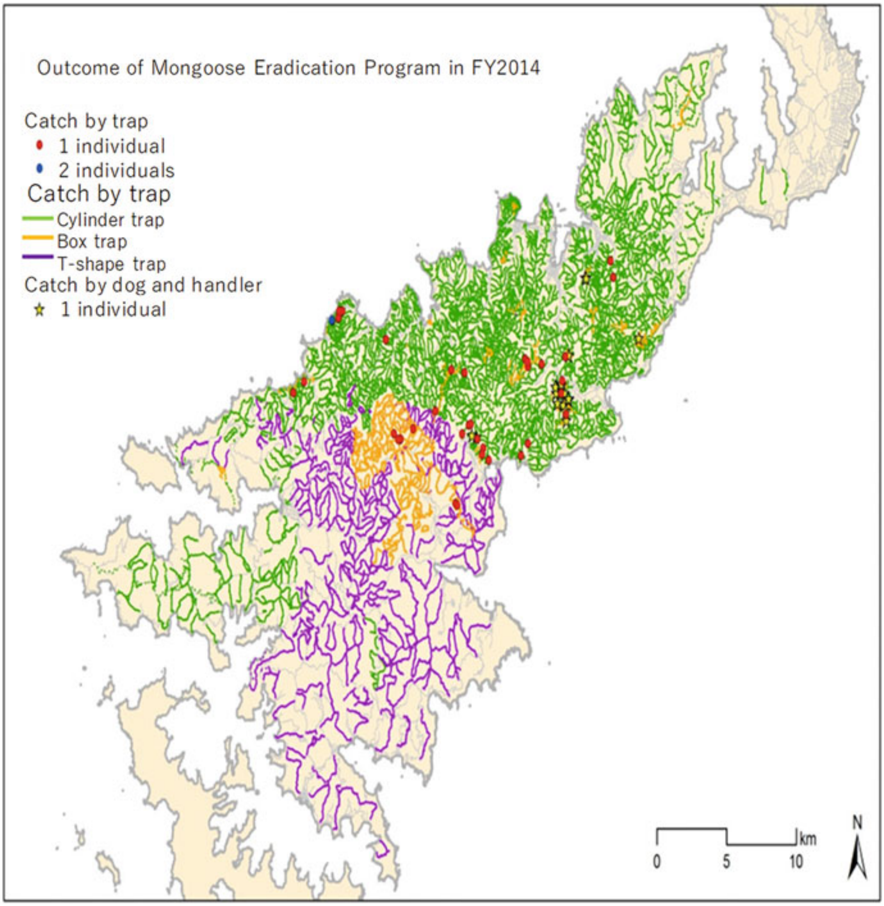

Fig. 12.7 Capture locations of mongoose by capture means in the fiscal year 2014. A total of 2.6 million trap days were set along the trap installation route (Ministry of the Environment, Japan 2015)

considering issuing an eradication declaration if there is no capture or observation as it is in the 2020s (Ministry of the Environment, Japan 2019).

12.7 Conservation Activities for the Paradise Fish in Okinoerabujima Island

There are activities to conserve a local population that may be non-native species. There is a freshwater paradise fish (*Macropodus opercularis*) on Okinoerabujima Island in Kagoshima Prefecture, Japan. Unlike the goldfish (*Carassius auratus*) of the carp, it is a genus *Macropodus* of the perch and is widely bred as a tropical fish for appreciation. Males have a territory and have the property of attacking other individuals that enter the territory, and males are often bred to fight each other for a show.

This fish used to be common in and around the paddy fields on the islands, but with the decrease in the number of paddy fields, the number of wild populations is decreasing. An NPO's activity "Project for homeland of paradise fish in Zikkyonuho, which connects children" was selected as one of the "Heritage for the Future" Movements in 2017 by a nongovernmental organization "the National Federation of UNESCO Associations in Japan."

In Japan, wild populations of paradise fish distribute in Ishigaki Island, Okinawa Island, Tokashiki Island, Kumejima Island in Okinawa Prefecture, and Okinoerabujima Island in Kagoshima Prefecture in Japan. It is uncertain whether these Okinawan populations were introduced before the Edo period or is a native species. Mitochondrial DNA of Okinawan populations is almost the same as populations of Taiwan and parts of the southern part of China (Yamashita and Watanabe 2018). It is highly likely that the paradise fish was artificially introduced into Okinawa in old times (Kano et al. 2018). The population of Okinoerabujima Island had inhabited 80 years ago and became part of the island's biota. On Okinoerabujima Island, it is highly likely that it is an established exotic species.

Japan's Ministry of the Environment listed the paradise fish as an endangered species of the Japanese Redlist. Outside the country, it is distributed in Taiwan, China, Southeast Asia, and so on. It is a common species as a whole including Southeast Asia. However, in Taiwan, as in Japan, it is endangered and has been designated by Taiwan's Wildlife Conservation Act.

According to Kano et al. (2018), rice farming in Okinoerabujima has been eradicated due to the rice acreage reduction policy since 1970, and paddy fields were transferred to sugar cane fields. The presence of paradise fish is an indicator species of wetlands that inhabit the paddy fields, while no particular significant negative effects by "exotic" paradise fish on native ecosystems are known. It may also play a role in reintroducing into Taiwan if the Taiwanese population becomes extinct, and the cultural value of "Fighting fish" in Okinoerabujima Island is respected. Therefore, it is considered a conservation target even among conservation ecologists.

The "Heritage for the Future" movement is not a UNESCO's project, but a system recognized by the National Federation of UNESCO Associations in Japan, which supports UNESCO. The "Heritage for the Future" Movement will shed light on the rich gifts of cultural heritage that people have continued to spin for a long time, and the rich heritage created through wisdom and ingenuity for living harmony with nature. A total of 73 cases were certified from 2009 to 2019 with the aim of supporting the citizens' activities and activating the willingness of people to be passed on to future generations.

Japan became a member state of the United Nations in 1956 after Japan joined UNESCO in 1951. There were private organizations called UNESCO Associations in various parts of Japan prior to 1951. This movement formed the current

Federation of UNESCO Associations in Japan. It is also a characteristic of Japan that private organizations with many local branch offices have supported the UNESCO movement for a long time.

The conservation activities of the paradise fish on Okinoerabujima Island are part of the activities to restore the tradition of the region around the spring pond called Zikkyonuho, which was the base of the village, and an elementary school promotes environmental education as the core of the conservation activities of the fish.

12.8 Cat Controversy Over Free-Ranging Cats

The treatment of exotic species is linked to the treatment of pets, livestock, and wildlife. A typical example is a domestic cat, which is treated by both pets and exotics. The domestic cat is an alien predator in the "100 of the World's Worst Invasive Alien Species" (Marra and Santella 2016). Domestic cats are said to have contributed, partly or mainly, to the extinction of 33 of the 238 extinct reptiles, birds, and mammals in the world.

Unlike dogs, cats are often unconnected, even if the owner keeps them. They often prey on field food themselves, even without their owner's knowledge. There are cats that have no owners of cats, cats that have owners and are left alone, and cats that are kept indoors only. The former two are called "free-ranging cats" (Marra and Santella 2016). The last two are continuous, some semi-reared cats and some cats in the colony where multiple persons feed. It is estimated that free-ranging cats and cats with their owners prey on 8.7–21.8 and 177.3–299.5 small mammals per year, respectively.

In addition, cats carry zoonotic diseases, including plague and rabies. Toxoplasma, which is sexually reproduced only in the intestinal tract of Feline, has long been known to pose a significant risk to the fetus when a mother is first infected during pregnancy. The prevalence of Toxoplasma (in women who can give birth) is 4% in South Korea and 63% in Germany (Marra and Santella 2016).

Since free-ranging cats prey on wild birds and mammals, there is a serious conflict between cat and bird advocates. It is known that nature conservation and animal welfare are different. Personal extermination of cats to protect wild birds may be accused of animal cruelty in the United States. On the other hand, the Endangered Species Act may condemn the owner for killing an endangered animal by a free-range cat. Free-ranging cats are legally vague, which are either pets or invasive alien species. The lack of political agreement on how to handle free-ranging cats is a problem in the world (Marra and Santella 2016).

In the United States, TNR (trap-neuter-return) of wild cats is recommended. However, there are criticisms that TNR does not help a solution (Marra and Santella 2016). It is not practical to achieve a level of sterilization that reduces the number of free-ranging cats. Not just cats, releasing exotic animals or any pets into the field is a disturbing factor in the ecosystem.

However, it may be controversial to claim that euthanasia of cats is ethical because wild cats live poorly. The “desirable world for birds, people, and cats” has been proposed in various ways. It is advisable to keep cats indoors and to attach a towline when owners take cats outdoors. Breeding should be allowed provided that the cat is bred with a microchip (Marra and Santella 2016).

Even if the cat is captured, whether to kill it is another matter. It can be transferred to an individual or housed in a special facility called a sanctuary, but there is a limit in terms of the number of cats and the capacity to receive them (Marra and Santella 2016).

Measures against alien species will also depend on the conditions. Free range should be stopped in ecosystems with endemic subspecies of small islands or in Australia and New Zealand where there was no similar natural enemy originally. However, free-ranging cats may not necessarily be eradicated in the main islands of Japan.

In this chapter, we explained the population ecology and bioeconomics of exotic species control. Eradication is not easy just by increasing the capture effort by traps. Our ultimate goal is not to eradicate exotic species but to conserve native ecosystems. If it can be suppressed to a low level with relative ease, the decision to continue making capture efforts forever is often reasonable. If we aim to eradicate, we have strong leadership and belief that we can deal with unexpected situations without upset and hesitation, know the distribution area accurately, hit the high-density area surely, and prevent the expansion of the distribution. It is important to develop a technique to effectively capture even low density and wait for the opportunity to extinct persistently because the final extinction event is triggered by demographic stochasticity.

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Chapter 13
Beyond Dichotomy in the Protection and Management of Marine Mammals

Hiroyuki Matsuda

Abstract Marine mammals are natural resources and some are considered as pest animals. The main factors driving the relationship between humans and marine mammals changed from the mid-twentieth century to the early twenty-first century. This is the result of changes in their extinction risk, resource demand, and animal welfare for wildlife. In this chapter, we selected Steller sea lions to investigate changes in Japanese marine mammal policies. Japan's policy of Steller sea lion has changed from resource utilization in the mid-twentieth century to conservation in the second half of the twentieth century and pest control since 2014. Japanese environmental groups have played an important role in building consensus on these policy changes. We call for a comprehensive policy that implements a balanced approach to address the three different roles of marine mammals: natural resources, pest animals, and targets of animal welfare. We also discuss the importance of stakeholder involvement in aiming for population management that is neither overfishing nor full protection.

Keywords Potential biological removal · Adaptive wildlife management · Dugong · Harbor seals · Population control · Stakeholder involvement · Steller sea lion

13.1 Population Structure and Trends of Steller's Sea Lions

Marine mammals provide ecosystem services, including food, fur, oil, and others (Pompa et al. 2011). In terrestrial ecosystems, mammals occupy a variety of niches in the food chain. In a marine ecosystem, marine mammals eat phytoplankton, zooplankton, invertebrates, fish, birds, or other mammals. Baleen whales often eat both plankton and small fish. Overall, they consume more fish than are caught by fisheries (Tamura et al. 1998; Yodzis 1988). In addition, marine mammals often conflict with fisheries. Traditional Japanese eat a variety of fish, invertebrates,

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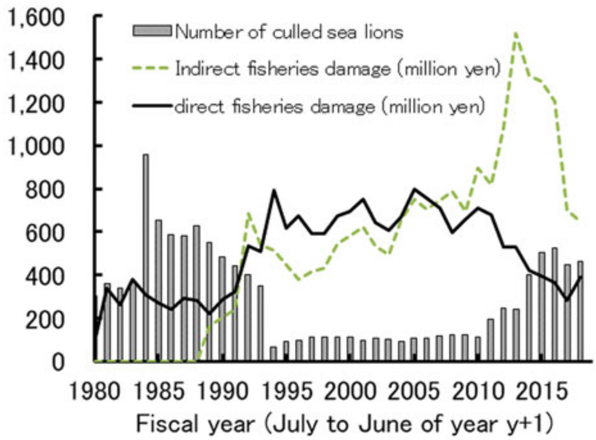

Fig. 13.1 The catch of Steller sea lions and the damage to fisheries in Japan (data from Fisheries Agency, Japan, see also Matsuda et al. 2015)

seaweeds, and even marine mammals. On Ishigaki Island in Okinawa, Japan, the Ryukyu Kingdom made people pay a tax on dugong (*Dugong dugon*) (Hosono et al. 2009). After the end of the Ryukyu government, dugongs were used as resources and were overfished. Since 2006, only three individual people were found around the main island of Okinawa (Okinawa Defense Bureau 2012), and these three disappeared or died by January 2019.

On the other hand, Steller sea lions (*Eumetopias jubatus*), harbor seals (*Phoca vitulina stejnegeri*), and spotted seals (*Phoca largha*) damage the gillnets and eat fish in the nets in Hokkaido and Aomori prefectures and are considered to be pest animal in the fishery. Hokkaido Prefecture has been compiled direct and indirect damage statistics on fisheries by sea lions since 1988. Direct damage refers to damage to fishing nets such as by attacking fish that are caught in the nets, and indirect damage is the sum of lost fishing opportunities and damage to caught fish such as salmons. Direct and indirect damage to fisheries by sea lions has increased since the 1990s. The direct damage to the fishery by sea lions was ca. 270 million yen in 1990 and ca. 490 million yen in 2012, as shown in Fig. 13.1. The indirect damage was about 190 million yen in 1990 and about 1 billion yen in 2012. However, increasing fishing opportunities may reduce future resources, so the indirect damage from the early closure of the fishing season is exactly unknown. When Fisheries Agency started the population control of sea lions since 2014, the indirect and direct damage were considerably reduced.

The Fisheries Agency, Japan, also considers whales as pest animals to fisheries. Although no direct damage of whales to fisheries is known, increased whale populations may reduce fish stocks (Tamura et al. 1998). This is one of the reasons the Fisheries Agency insisted on the need to resume commercial whaling because of the conflict between fisheries and whales. However, the scientific debate about the interaction between whales and fisheries resources is controversial (Yodzis 1988).

Steller Sea Lion (*Eumetopias jubatus*) distributes in the northern Pacific Ocean from Japan to California, whose population size is ca. 80,000 in 2015. This species is

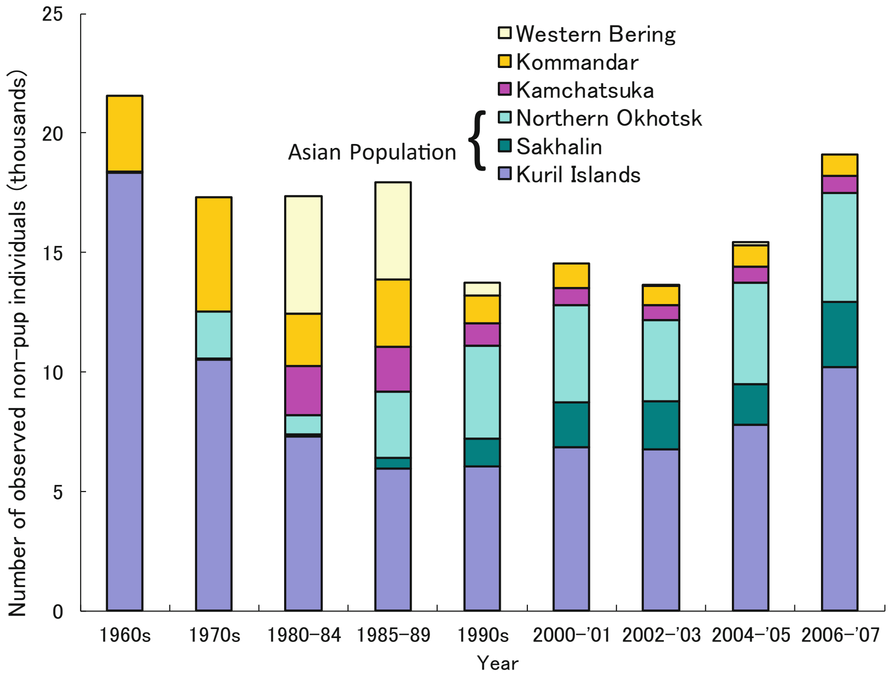

Fig. 13.2 Population trends of Western Steller Sea Lion (based on the data from Burkanov and Loughlin 2005; Burkanov et al. 2008, Hattori and Yamamura 2010)

divided into two subspecies, the Western Steller Sea Lion (*E. j. jubatus*) and the Loghlin's Steller Sea Lion (*E. j. monteriensis*) (Phillips et al. 2009). Distribution of these subspecies is divided at 144°W longitude. The eastern subspecies is increasing, while the western subspecies decreased until the end of the twentieth century. Eastern and western subspecies are Least Concerned and Endangered in IUCN's Red List, assessed in 2015, respectively, whereas the whole species is near-threatened (Gelatt and Sweeney 2016).

The western subspecies can be divided into "Central" and "Asian" populations (Baker et al. 2005), although these two populations have no significant genetic difference from each other (Hoffman et al. 2006). Asian population had sharply decreased until around 1990, as shown in Fig. 13.2. We estimated that the extinction risk half a century later would be about 20% if the rate continued to decrease.

Fortunately, Asian population began to recover gradually since 2000. For this reason, the Ministry of the Environment (Japan) has designated this species as Vulnerable in the Japanese Red List (Matsuda et al. 2015). As a result, domestic public opinion to reconsider the quota was strengthened. However, the criticism of extermination of sea lions from abroad has not diminished. Considering that Shiretoko is inscribed as a World Heritage and that sea lions have been eradicated in Shiretoko, the review of catch quotas may be subject to international criticism.

13.2 Population Control of Harbor Seals in Cape Erimo, Hokkaido, Japan

The harbor seal is not threatened internationally, but it has an isolated regional population at Cape Erimo in Japan and was listed as Endangered in the Japanese Red List until 2012. In addition, another population is distributed near the border between Japan and Russia. The Erimo population is said to have decreased to 350 in 1980s, but it has increased since then, and the damage to the fishery became serious. Harbor seal is downgraded from Endangered to Vulnerable in the Japanese Red List in 2012. Local fishers petitioned the Ministry of the Environment for measures to fisheries damage. The director of the Natural Environment Bureau of the Ministry of the Environment replied to take measures such as capture. However, the Minister of the Environment overturned it, which caused a backlash from fishers.

After that, the review work of Japan's Red List proceeded quickly, and in 2015, the harbor seal was again downgraded to near-threatened. At that time, IUCN's criterion D regarding the population size was not applied, and the reason for the downgrade was that it did not meet criterion E regarding to extinction risk. In parallel with this process, Japan's Protection and Control of Wild Birds and Mammals and Hunting Management Law was enforced in 2015, and the wild birds and mammals management plan is divided into three types; type I is a protection plan that prevents further decrease in population size, and type II is population control to decrease human-wildlife conflict; type III is specified rare wildlife management plan which targets a threatened species but also requires reduction of human-wildlife conflict. Of these, the former two will be formulated by prefectures, and type III will be managed by the Ministry of the Environment. The only species to which the type III management plan was applied was the Erimo population of harbor seal, which had been downgraded to near-threatened.

There is no rule that population control should not be applied to endangered species. This point may have been misunderstood by the Minister of the Environment. The Red List does not limit policy, it is based on the precautionary principle and is determined regardless of the actual political situation. In the Japanese Plant Red List as mentioned in Chap. 8 and Marine Organism Red List, species that do not meet Criterion E are not to be listed on the Red List even if they meet other criteria. A species whose extinction risk is unknown is to be listed by other criteria. Harbor seals have been downgraded to endangered species, usually by accelerating the review schedule every 5 years. If it were a measure to implement population adjustment as soon as possible, it would be a double fallacy.

Steller sea lions and cetaceans are managed by the Fisheries Agency of Japan as pest marine mammals to fishery or fisheries resources, while other marine mammals including harbor seals are managed by the Ministry of the Environment. The Ministry of the Environment, Japan, has set up a new office in Erimo's fishing village and assigned a ranger. The first ranger has succeeded in building the trust of local fishers, and a harbor seal management program is underway with the consent of the fishers. A small number of isolated populations maintain a population that does not meet Red List Criterion E, which states that the extinction risk over the next one century will be less than 10%, even considering the risk of infectious diseases.

The government decided to control the population size to reduce the damage to the fishery. A scientific committee has been established that includes fishers with local knowledge, as well as fisheries scientists, ecologists, and mammalogists. Based on the recommendations by the science committee, the government formulated a management plan every 5 years through the procedure of public comments. In the annual action plan, the Ministry of the Environment sets catch quotas based on the advice of the scientific committee and the opinions of local briefings. Similar management systems for wildlife and World Natural Heritage is established in Japan.

13.3 Potential Biological Removal for Steller's Sea Lion

Public attention to the protection of Steller sea lions is increasing internationally. There has been criticism of Japan as it continues to cull this species even near Shiretoko World Heritage site. As a result, the culled number of sea lions was limited to 116 per year since 1994. The missing and injured individuals were included in this limit. There is no scientific reason that the cull limit of 116 guarantees the population persistence.

For this reason, the Fisheries Agency, Japan decided to apply the cull limit based on "potential biological removal" (PBR) used by the National Oceanic and Atmospheric Administration (NOAA) (Matsuda et al. 2015). PBR means that even if there is a human-caused mortality equivalent to half the rate of natural population increase, the population size can be expected to maintain at more than half of the carrying capacity. With the number of deaths as the upper limit, the measurement error of the population size is taken into consideration, and human-caused mortality is limited for endangered species to recover more quickly (Wade 1998). For example, 10% of the above-calculated value for Endangered or Critically Endangered species and 50% for Vulnerable species are set as the allowable limit of human-caused mortality.

Annual limit of human-caused mortality per year based on PBR is defined as $N_{min}R_{max}F_r/2$, where N_{min} is a conservative estimate of population size; R_{max} is the maximum population growth rate per year, and F_r is a recovery factor that is 0.1 for Endangered species, 0.5 for Vulnerable species, and 1 for common species (Wade, 1998). According to the Working Group for Stock Assessment of Steller Sea Lions (hereafter abbreviated by WG-SSL), $N_{min} = 5063$ from aerial surveys; $R_{max} = 0.12$ and $F_r = 0.75$ because the sea lions were classified as Vulnerable in the Japanese Redlist and their population was increasing. As a result, the WG-SSL determined that the upper limit of human deaths was set at 227.

In practice, there is not only catch but also bycatch in fisheries. This must be included in human-caused mortality. However, there is no legal reporting requirement for bycatch, and the actual number of bycatches is unknown. If the cull limit increased under the assumption of no bycatch, it would be accused internationally of misusing the PBR concept. The WG-SSL guessed that the number of bycatches was around 106 (Hattori and Yamamura, 2014), and the catch limit was set to 120 (Matsuda et al. 2015), which means the catch limit increased by four sea lions.

Fishers would have been disappointed but clarifying the facts about bycatch and showing that bycatch numbers are lower could increase catch quotas. It will motivate accurate reporting of bycatch numbers.

The key drivers of the relationship between humans and marine mammals have changed from the mid-twentiethth century to the early twenty-first century. This is due to the degree of population threat, the demand for these natural resources, and the changing policies for marine mammals. Marine mammal policies have changed from resource utilization in the mid-twentieth century to conservation in the late twentieth century to nuisance control since the 2010s. Matsuda et al. (2015) sought a comprehensive policy that achieves a balanced approach that addresses three distinct roles of marine mammals: natural resources, components of marine ecosystems, and drivers of damage to fisheries (Lavigne 2003). They also discussed how consensus can be reached in changing wildlife management strategies for these marine mammals compared to seals and sea lions.

Steller sea lion meat is on the menu of a restaurant in Rausu Town, adjacent to the Shiretoko World Heritage Site (Fig. 13.3). The serving size is small, and 1000 meals are available from one culled sea lion, but it seems that less than a hundred customers ordered this menu per year. Therefore, if one sea lion is culled in Shiretoko, the meat sold at this restaurant could be supplied.

According to the report of reactive monitoring mission team (UNESCO and IUCN 2008) strongly discouraged to cull or to use sea lions. Prohibition of eating sea lions will not stop killing but is concerned to discourage bycatch reports.

It is said that the origin of sea lion hunting on Hokkaido dates back to the Okhotsk culture about a 1000 years ago. Sea lion hunting is a "culture" that has been inherited by ethnic minorities called Nivkh and Uilta and local fishers, and the technique of collecting sea lions on a ship without sinking into the sea is an "intangible cultural property." After the inscription of Shiretoko World Natural Heritage, the restaurants that served sea lions in Rausu disappeared (Ohtaishi and Honma 2008).

Apart from capture by a single occupational hunter, sea lions have been culled to reduce damage to fisheries. On the other hand, there is no empirical or scientific evidence that eliminating sea lions reduces fisheries damage unless they are completely eradicated. However, if sea lion culling is banned, fishers will feel denied their fisheries. For this reason, the cull number of sea lions is limited to reach a compromise between the fishers' benefit and protection of sea lions.

One culture disappeared from the Shiretoko World Heritage Site. It may not be necessary for the conservation of the sea lions. If a sea lion is economically valuable, it will not be just a pest animal. If people catch sea lions, sea lions may fear people and stay away. Similar to the case of bears in Chap. 13, the "sustainable tension relationship between humans and wildlife" leads to coexistence. Cultural diversity is acknowledged in the Convention on Biological Diversity. People may consider diversity in human values to be more important than biodiversity.

Fig. 13.3 A menu of Steller sea lion's meat of a restaurant in Rausu (Photo by Hiroaki Honma, Mainichi Shimbun)

13.4 Population Control of Steller's Sea Lion in Japan

The Fisheries Agency has set an upper limit on the number of catches by PBR and has not reduced the number of sea lions. However, in 2012, the sea lions were downlisted from vulnerable to near-threatened, and population control began in 2014.

The Basic Plan for Fisheries states that "Ensuring the development of fisheries in the coexistence of various marine life," in the sea area where sea lions come. We aim to (1) minimize damage on fisheries caused by sea lions within the population level whose extinction level is negligible and to (2) adopt an adaptive management plan with precautionary measures, recognizing that the population of sea lions were once overexploited and faced on the threat of extinction. In addition, to reduce damage, the management goal was to reduce the number of migratory sea lions 10 years later (FY2024) to 60% of 2010. The Fisheries Agency organized the scientific committee to review this basic management plan. A member of an environmental organization was invited as one of the committee members.

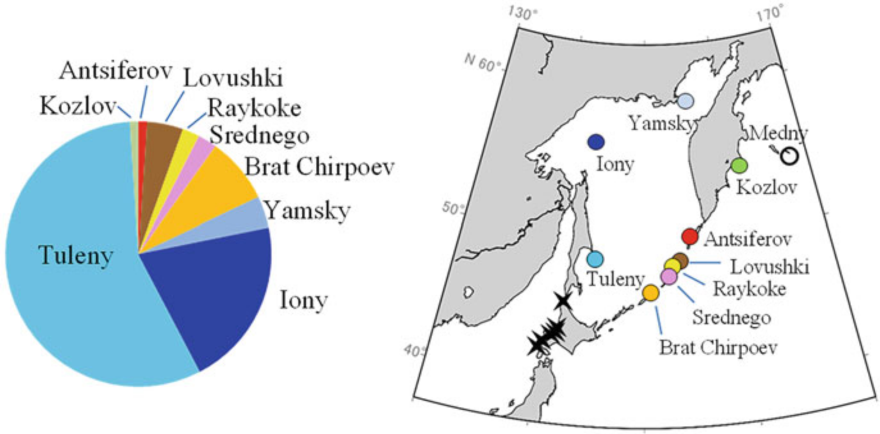

Fig. 13.4 (left) Frequency distribution of sea lions by breeding sites found in landing sites in Hokkaido excluding the Sea of Okhotsk. (right) Distribution of birth breeding grounds and observation sites in the Asian region. “X”s mean major landing sites in Japan (source: Fisheries Research and Education Agency 2017).

The management plan agreed in 2014 and its revised plan in 2019 only covers the Sea of Japan. A management plan should be made for the sea lions coming to Shiretoko, the Sea of Okhotsk too. There are three reasons.

First, the Fisheries Agency has a responsibility to manage not only the Sea of Japan but also the sea lions in the Sea of Okhotsk, including Shiretoko, based on scientific evidence. The Fisheries Agency should deal with the damage to the Shiretoko fishery. Even if there is little scientific knowledge and it is not possible to accurately estimate the number of migrants to the Sea of Okhotsk, it is possible to set a catching quota with a pessimistic estimate. In Shiretoko, the number of catches may be larger than the number of migrants in recent years, but this is a matter of allocation of catch quotas in Hokkaido, and this is not a reason for not carrying out scientific management of sea lions in the Sea of Okhotsk (Fig. 13.4).

Second, they are not separate populations between the Sea of Japan and the Sea of Okhotsk. Certainly, birthplaces are divided into Sakhalin Island and Kuril Islands, and most of the sea lions that come to Shiretoko come from Kuril. However, more than 10% of the sea lions coming to the Sea of Japan not only come from the Sakhalin side such as Chuleny Island or Iony Island, but also from Kuril Islands such as Brattilpoev Island (Fig. 13.3). If 500 sea lions are caught in the Sea of Japan, more migrants from the Kuril island are caught in the Sea of Okhotsk than the catch quota in the Sea of Okhotsk. In addition, it is not always necessary to distinguish between the lions breeding on Chuleny Island and Sakhalin because not a few individuals migrate from Kuril to Sakhalin.

Third, when the PBR was set, the cull limit included Shiretoko. We explained that when Shiretoko was inscribed as a World Natural Heritage site in 2005, the cull limit prevented further decline in the population. The current cull limit in Shiretoko lacks such a scientific reason. The World Heritage Committee has repeatedly asked the legitimacy of killing sea lions. The Fisheries Agency has refused to be a party to the Shiretoko World Natural Heritage Management Plan. The Shiretoko fishers voluntary expanded the seasonal fishing ban zone during the registration of the Shiretoko World Heritage Site (Makino et al. 2009). This was highly regarded as being selected as one of the six "World Impact Story" by the International Association for the Studies of the Commons in 2010. An officer of The Ministry of the Environment, not the Fisheries Agency, persuaded the fishermen at that time. This is in sharp contrast to the Forestry Agency's participation in the Deer Management Plan and in the River Structure Working Group of Shiretoko World Heritage Site. I suspected that the agency tried to escape international condemnation by cutting Shiretoko from the cull limit.

Finally, when reviewing the management plan, the opinions of environmental organizations should be fully respected. As mentioned above, WWF Japan participated in the Scientific Committee and gained social consensus when the management plan was formulated, and the cull limit was greatly increased in 2014. Excluding them from the consensus-building debate would not gain an international understanding.

We began the population control plan of Steller sea lion in 2014, but there was no guarantee that this would be successful. First, the management plan assumes that sea lions that come to Japan are an isolated population, but in fact a part of the Asian population comes to Japan. Exterminating in Japan does not mean that the number of migrants to Japan will not decrease and that individuals who did not come to Japan last year may come. Another possibility is that surviving individuals and their descendants may avoid Japan, and the number of migrants will decrease more than we culled. Second, the amount of damage may not be proportional to the number of migrants. As discussed in the chapter on bears, there may be individuals who eat fish in the set net and those who avoid humans.

As of 2018 when the first 5-year plan ended, the number of sea lions coming to Japan did not significantly decrease. This is a stage where the number of catches is not sufficiently large and no significant change can be expected, and success or failure cannot be evaluated. However, there were signs that the distribution within Japan had changed significantly. Until now, many fisheries have been damaged from Cape Soya to Aomori Prefecture, but most individuals who came to Cape Soya remain at Bentenjima, a reef near Cape Soya (Fig. 13.5). As a result, indirect damage to fisheries has been significantly reduced, though the damage around Cape Soya was still serious. This effect was not described in the management plan in 2014. As the distribution pattern of sea lions is still changing, we need to look at the progress a bit more. We will need to consider how to change the victims by the population control and to prepare the countermeasures.

Fig. 13.5 A large herd of sea lions landing on Bentenjima in Soya Straight, Japan (photo by Wakkanai Fisheries Experimental Station on February 27, 2018). More than 3000 sea lions landed on the island with an area of 50a

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Chapter 14
Management of Human–Bear Conflict

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Abstract On Oshima Peninsula, Hokkaido, Japan, recent trends concerning the intrusions of the brown bear into crop fields and a subsequent increase in agricultural damage have highlighted the need for new and more effective population management strategies. To devise such strategies, we constructed a population dynamics model for adult females, based on bear aggressiveness and human-bear interactions. The results of the analysis indicate that an adaptive management strategy successfully reduces the risk of management failure. In Shiretoko World Heritage site, also Hokkaido, wild bears become a nuisance by interacting with photographers who visit the area. Through such interactions, bears get used to the presence of people, and due to a lack of fear, may eventually exhibit aggressive behavior. Managing bear populations, therefore, also demands managing people. To further exacerbate the situation, the human–wildlife conflict has spilled from national parks to metropolitan areas. Ban on hunting does not help either.

The human–wildlife conflict is common in both developing and developed countries. African elephant ivory has historically achieved a high price that subsequently caused a dramatic fall in the population between 1979–1989. Elephants were listed in CITES Appendix I in 1989. Populations in southern Africa have since rebounded and, despite an increase in poaching and smuggling after the ban on exports, they have been downgraded to Appendix II. It is not the case that humans exclusively threaten elephants. In Tanzania, for example, dozens of people annually get mauled by elephants, with the human–elephant conflict generally being more critical around nature reserves. In Europe and the United States, thriving trophy hunting turned out to be a powerful means of managing damage caused by animals. Developing countries sometimes seek to mimic this positive effect, for example, by game hunting for elephants and tigers.

Wild animal meat (bushmeat), fish, shellfish, and even insects are important protein sources, especially in poorer areas of the world, such as sub-Saharan Africa.

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Therefore, there is no alternative to making use of wildlife for food more sustainable, while managing the human–wildlife conflict. The same holds for farmland, which is usually the land that had once been occupied by wild birds or beasts before being taken over by human inhabitants. If farming is to coexists with wild birds and beasts, some level of the human–wildlife conflict is inevitable in agriculture, as well as in forestry and fisheries. People are inseparable from the biosphere, and thus not only use wildlife but are sometimes being used by wildlife too. Under the Convention on Biological Diversity, the basis of nature conservation is that there are ecosystem services or nature's contribution to people that promote human well-being. This leads to the idea of Sustainable Development Goals.

Keywords Ecological risk · Population dynamics · Food conditioning · Human–wildlife conflict · Nuisance bears · Biosphere

14.1 Coexistence with Bears that Attack People

Land development and other human activities limited wildlife's habitat in Japan in the 1970s. Nowadays, those habitats are expanding due to the aging and declining human population in the mountainous areas. There are two species of bears in Japan, Brown bear (*Ursus arctos*) and Asiatic black bear (*Ursus thibetanus japonicus*), that are listed in the CITES Appendix I and II, respectively. Brown bears are present in Hokkaido Prefecture, while black bears are present in Honshu and Shikoku islands and a number of smaller, satellite islands. Black bears furthermore went locally extinct in Kyushu Island. The two subpopulations of brown bears, Shakotan-Eniwa and Teshio-Mashike, are listed as "threatened local populations" by the Ministry of the Environment.

Populations of brown and black bears are likely to be increasing, but there is no quantitative scientific evidence for population sizes or the rates of population increases. The number of human–bears conflicts, however, is well-documented, and instances of bears attacking people or damaging agricultural farms are certainly on the rise. Assuming that the number of conflicts is roughly in proportion to the population size of bears, further increases in the frequency of incidents is likely to garner public support for enacting population control measures. The conflict cannot be eliminated as long as wild animals are present, but it can be kept within an acceptable level. If the number of conflicts does not depend on the population size of bears, population control measures are clearly not an answer, and other means of managing the human–bear conflict will have to be contemplated.

The brown bear population has historically been known to cause injuries and agricultural damage. From 1988 to 2005, the number of bears killed increased by 6.7% annually, yet the amount of crop damage during the same period also increased by more than 5% annually (Mano, 2009). Therefore, bear killing in Hokkaido may have failed to prevent wild bears from entering farm fields and achieved little in terms of food conditioning as defined by Gunther and Wyman (2008).

14.2 "Susceptible" Bears and "Nuisance" Bears

How is it possible that both the number of killings and the number of incidents are simultaneously on the rise? There are two types of bear individuals, those that cause conflicts and those that do not, with the former increasing and the latter decreasing in the population. In the Ainu language, the brown bear is called "ki-mun-kamuy," which means "god in the mountain" (Ohta et al. 2012). On the other hand, bears that cause incidents are called "wen-kamuy," which means "bad god." This traditional classification of bear individuals roughly corresponds to a more scientific division into susceptible and nuisance bears, respectively. Ki-mun-kamuy spend time mostly away from human settlements, and in this sense are mostly, but not entirely, harmless to people. These bears are, however, "susceptible" to adopting the behaviors of nuisance bears, where the term "susceptible" is used in analogy to the use of this term in infectious-disease modeling. We consider that a "susceptible" bear turns into a nuisance bear by improper conditioning such as feeding on farmland. Recovering to a susceptible state is then possible by aversive conditioning. Importantly, improper conditioning helps wild animals overcome the fear of humans, meaning that bears subjected to such conditioning no longer avoid humans. The distance between humans and wild animals thus decreases and conflicts can no longer be avoided.

In order for people and wild bears to coexist, and for the number of conflicts to reduce, measures should be taken to elevate the proportion of susceptible bears relative to nuisance bears. Previously, mammalogists assumed that susceptible bears lived in the mountains and nuisance bears lived near settlements. Recent studies, however, suggest that some bears living near settlements carefully avoid people, and therefore do not cause conflicts. It is thus more appropriate to distinguish susceptible and nuisance bears by behavior rather than their habitat. This furthermore has implications for aversive conditioning, which should be applied in the bear's natural habitat, instead of attempting to translocate nuisance bears to mountainous areas.

The suggested classification of bears into two types cannot be effective in mathematical ecology models if it is not based on practical standards for distinguishing susceptible from nuisance individuals. Mammalogists in Hokkaido accordingly developed a method to distinguish these two bear types and to estimate the number of susceptible and nuisance individuals. There is now a "diagnostic guide" for nuisance bears in place in order to make the right judgments based on information available in bear-sighting reports. Just because people say they saw a bear it does not mean that they spotted an actual nuisance bear. Many bears do not cause problems even if they happen to encounter people. It is, in fact, bears who learn to easily get food in farmland that exhibit a lack of fear, and even attack people for food. Such fearless bears are the true nuisance.

Beyond just distinguishing susceptible from nuisance bears, it is also necessary to understand whether various reports refer to one or more individuals in the neighboring area. To this end, attempts have been made to perform DNA analyses using a device called the "hair trap." As the name suggests, the device collects hairs from the fur of bears that enter farmland fences. The results show that both possibilities get realized; sometimes multiple bears enter the same observation site, and sometimes a single bear enters adjacent sites. Although the budget and personnel for DNA testing using hair traps are limited, valuable information on the proportion of nuisance bears in the population can be extracted.

14.3 Mass Appearance of Bears

Bears sometimes appear in large numbers in people's settlements, increasing the instances of conflict; we call this "mass appearance." The number of bear catches also increases in such years. The reason for mass appearances is that bears lack food in their natural habitat. Female bears hibernate in winter and give birth to cubs during hibernation, implying that they need to accumulate substantial reserves in autumn. The staple food during this period is beech (*Fagus crenata*) and Japanese oak (*Quercus crispula*), yet the yield of these nuts varies greatly depending on the year. Trees in the mountains, in fact, exhibit a curious synchronization phenomenon that results in periodic booms and busts. Booms naturally result in many cub births, while busts preclude reproduction. Adults rarely starve to death. These natural trends notwithstanding, bears experiencing food scarcity seek to feed themselves near human settlements, resulting in above-average culling.

When assessing the extinction risk of wildlife species, it is difficult to make judgments based on limited data. For example, if we only looked at the high-feeding years, the data will point to very high reproductive rates. Assuming thereafter that such reproductive rates are normal every year, the bear population may be falsely deemed safe even if many individuals get culled. Conversely, if we only looked at the low-feeding years, the data will point to very low reproductive rates thus prompting worries about the population's sustainability even without culling. None of this would be true, of course. The established picture nowadays is that the number of bears fluctuates due to the mixture of years with high and low food abundance.

14.4 Population Dynamics Using Matrix Population Models

If the population size were characterized by one variable (i.e., a scalar), the future increase or decrease of the population could be represented by the geometric mean of the population growth rates. To illustrate this, we denote the number of individuals in year t by $N(t)$ and the population growth rate in year t by $e^{r(t)}$. The dynamical equation is (Ohta et al. 2012):

$$N(t+1) = e^{r(t)} N(t) \tag{14.1}$$

for any t. We obtain $N(t)$ by using $N(0)$:

$$N(t) = e^{\sum_{s=0}^{t-1} r(s)} N(0) \tag{14.2}$$

Therefore, if the arithmetic mean of the logarithm of the population growth rate, $r(t)$, is positive, the population increases, i.e., $N(t) > N(0)$. Otherwise the population decreases. This is equivalent to stating that, if the geometric mean of the population growth rate, $e^{r(t)}$, is greater than unity, the population increases, and otherwise it decreases.

The composition of the population, however, is rarely characterized by a scalar. Taking into account age or stage structure is often necessary, and fluctuating environments even further complicate matters. Considering as an example a hypothetical population dynamics comprising one-year-old children, $J(t)$, and two-year-or-older adults, $A(t)$, the dynamical equation becomes

$$\begin{pmatrix} J(t+1) \\ A(t+1) \end{pmatrix} = \begin{pmatrix} 0 & S_0(t)R(t) \\ S_1 & S \end{pmatrix} \begin{pmatrix} J(t) \\ A(t) \end{pmatrix}. \tag{14.3}$$

Using vector $\mathbf{N}(t)$ and matrix $\mathbf{L}(t)$ the equation turns into

$$\mathbf{N}(t+1) = \mathbf{L}(t).\mathbf{N}(t). \tag{14.4}$$

The quantity $R(t)$ is the number of offspring born by one adult female, $S_0(t)$ and S_1 are the survival rates from 0 to 1 years old and from 1 to 2 years old, respectively; S is the annual survival rate of adults. Here, we assume that S_1 and S are constant every year regardless of the environmental conditions, and $S_0(t)R(t)$ changes annually depending on the environment. In many cases, fecundity and mortality of infants are most strongly affected by environmental conditions.

For simplicity, we assume that $\mathbf{L}(t)$ takes one of the following two forms depending on how good or bad the environment is:

$$L_G = \begin{pmatrix} 0 & 2 \\ 0.2 & 0.8 \end{pmatrix}, L_B = \begin{pmatrix} 0 & 0 \\ 0.2 & 0.8 \end{pmatrix}, \tag{14.5}$$

In other words, two juveniles survive in a good year, but none in a bad year. If good and bad years alternate, the population after 10 years is

$$\mathbf{N}(10) = \mathbf{L}_B.\mathbf{L}_G.\mathbf{L}_B.\mathbf{L}_G.\mathbf{L}_B.\mathbf{L}_G.\mathbf{L}_B.\mathbf{L}_G.\mathbf{L}_B.\mathbf{L}_G.\mathbf{N}(0). \tag{14.6}$$

Because matrix multiplication is associative, the population becomes

$$\mathbf{N}(10) = (\mathbf{L}_B.\mathbf{L}_G)^5.\mathbf{N}(0) = \begin{pmatrix} 0 & 0 \\ 0.16 & 1.04 \end{pmatrix}^5 \begin{pmatrix} J(0) \\ A(0) \end{pmatrix}. \tag{14.7}$$

Therefore, the number of adults increases by 4% every 2 years and by 22% over the period of 10 years. In contrast, if there are 5 good years in a row followed by 5 bad years, the population is

$$\mathbf{N}(10) = \mathbf{L}_B{}^5.\mathbf{L}_G{}^5.\mathbf{N}(0) = \begin{pmatrix} 0 & 0 \\ 0.125 & 0.721 \end{pmatrix} \begin{pmatrix} J(0) \\ A(0) \end{pmatrix}, \tag{14.8}$$

meaning that the number of adults will decrease by 28% over the period of 10 years. Although in both cases there are 5 good and bad years in total, the population growth rate depends on the exact arrangement, alternating or continuous. Mathematically, the reason for this difference is that matrix multiplication is noncommutative. The described subtlety is not always appreciated by ecologists who, upon obtaining the average value of $S_0(t)R(t)$ through several years of observation, deduce the population trends by means of such an average. In our example, averaging results in the following matrix:

$$L = \begin{pmatrix} 0 & 1 \\ 0.2 & 0.8 \end{pmatrix}. \tag{14.9}$$

In this case, the number of individuals neither increases nor decreases.

Tuljapurkar (1989) first pointed out that the population trends depend on the autocorrelation in the environmental time series. Although noncommutative nature of matrix multiplication is well-known, it is relatively new for ecologists to pay attention to the effects of environmental change.

14.5 Adaptive Management of Nuisance Bears

If we remove nuisance bears and prevent susceptible bears from becoming nuisance bears, we can secure population sustainability, while alleviating the human–bear conflict. It is, however, difficult to exclusively select nuisance bears for culling. Uncertainty and mistakes cause susceptible bears to be culled. Furthermore, failing to cull nuisance bears who then reproduce, improves the chances of offspring also becoming nuisance bears by learning the behavior of their mother. Using statistical terms, culling a susceptible bear is a type I error while missing to cull a nuisance bear is a type II error. Management planning is a practical task that needs to account for the possibility of committing both type I and type II errors.

Black bears in Shikoku Island are threatened due to a decrease in habitat and population, while brown bears in Hokkaido and black bears in eastern Japan are not in any immediate threat. For populations with negligible extinction risks, there is little reason to avoid culling nuisance bears, despite type I errors, although it is important to keep the error rate as low as possible. The task of lowering the error rate is not only important but critical in cases when the managed population is also threatened, especially when the population size is less than a hundred, as is the case with black bears in Shikoku. It is also crucial to recognize that culling cannot be the answer to the rising bear–human conflict if susceptible individuals become nuisance bears due to people's improper behavior.

Key factors in bear management are the population size, the number of nuisance bears, the "incidence rate" (i.e., the rate of change of susceptible to nuisance bears), the number of captures, and the type I error rate. The estimation of bear population size is difficult, and in Hokkaido probably much less accurate than, for example, estimating the size of the deer population. Various periodic surveys may give some indication as to whether the population size is increasing or decreasing. The number of nuisance bears, as mentioned earlier, can be inferred to some extent by carefully monitoring the encounter rate and the magnitude of conflicts. The incidence rate is another difficult-to-estimate parameter whose accuracy cannot be known unless it is calculated using actual data. The number of captures is known, yet the diagnosis of the erroneous catch is important. In Hokkaido, there is now a guide to diagnose whether a caught bear should be considered a nuisance or not depending on that bear's behavior when it encountered people and whether there were agricultural products in the stomach contents.

The most desirable condition is that there are many susceptible and a few nuisance bears. In this case, there is a low risk of extinction of the population, type I errors are not critical, and game hunting of bears is possible. On the other hand, the situation in which the bear population is threatened and many nuisance bears exist is the worst. In this case, it is desirable to reduce the errors as much as possible and explore the possibility of keeping nuisance bears alive and subject to rehabilitation via aversive conditioning. In order to decrease the incidence rate, strict controls are needed over garbage disposal and access to crop fields, which can be prevented by installing electric fences. In emergencies, it is further possible to restrict tourist access to protected areas such as national parks.

The bear population on Oshima Peninsula in Hokkaido has been considered rich in the number of both susceptible and nuisance bears, demanding efforts to reduce the proportion of the latter bears in the population (Ohta et al. 2012). A surveillance system has therefore been put in place to monitor the number of nuisance bears, as well as that of erroneous catches. This has been done in response to convincing arguments by mathematical ecologists.

14.6 Population Dynamics Model with Management Measures

Female brown bears usually begin to reproduce at the age of 6 (Mano and Tsubota 2002). Supposing that the number of adult females in year t-6 determines the number of juveniles that begin reproductive activity in year t, Ohta et al. (2012) used for the average rate, denoted $r(t)$, the following equation

$$r(t) = Re^{[A(t-6)-\omega N(t-5)]} \tag{14.10}$$

where R is the average recruitment rate; $A(t)$ quantifies acorn production in the fall of year t; and ω is the strength of the density effect. Considering the experimental evidence suggesting a significant density effect (Czetwertynski et al. 2007), the average recruitment rate R is achieved only in the limit of a very small population. Also, the recruitment in year t depends on acorn production in autumn 6 years ago, denoted $A(t$-6), because the reproductive rate decreases with food shortage at the moment of reproduction.

It is known that acorn production strongly affects the recruitment of bears (Oka et al. 2004), as is known that the acorn production has a strong negative autocorrelation with a 2-year delay, which has been inferred from the dataset for Japanese oak (*Quercus crispula*) at Ogawa Forest Reserve in Ibaraki Prefecture between 1995 and 2004 (Shibata et al. 2019). Ohta et al. (2012) therefore used the equation

$$A(t+2) = \rho A(t) + \varepsilon(t) \tag{14.11}$$

where ρ and ε are respectively the coefficient of autocorrelation and the uniform random variable that satisfies $|\varepsilon(t)| \leq 1-\rho$. The population dynamics arising from the above considerations is

$$\begin{aligned} N_0(t+1) &= SN_0(t) + r(t)N_0(t-5) + \alpha(t)\beta C_1(t) - mN_0(t) \\ &\quad - [1-\alpha(t)]C_0(t), \\ N_1(t+1) &= SN_1(t) + r(t)N_1(t-5) - \alpha(t)\beta C_1(t) + mN_0(t) - [1-\alpha(t)]C_1(t). \end{aligned} \tag{14.12}$$

where $N_0(t)$ and $N_1(t)$ are the numbers of susceptible and nuisance female bears in year t, respectively. Here, the first two terms on the right-hand side of both equations are the number of surviving bears from year t-1 to year t, and the number of juveniles born in year t-5, which turn adults in year t. The third term in Eqs. (14.12) counts the number of nuisance bears that are recovered by aversive conditioning. Similarly, the fourth term in Eq. (14.12) counts the number of susceptible bears that become nuisance bears due to improper conditioning. Finally, the last term represents the number of killed bears, where $\alpha(t)$ is the release rate and $C_0(t)$ and $C_1(t)$ are random variables that represent the catch number of susceptible and nuisance female bears in year t, respectively. We ignore the number of male bears in this model.

Table 14.1 Parameter values used by Ohta et al. (2012)

Definition	Symbol	Value or range used in simulations
Average recruitment rate	R	0.17
Survival rate	S	0.95
(initial) release rate	$\alpha(t)$	(0 or) 0–0.8
Aversive conditioning success rate	β	0.3–0.6
Acorn production autocorrelation coeff.	ρ	−0.4
The initial (1987–1992) ratio of nuisance to susceptible bears	$N_0(t)/N_1(t)$	0–0.075
Initial population size relative to the harvest-based 95% CI during 1987–1992	ξ	0.017–0.819
False-catch coefficient	F	0.054–0.999
Incidence rate from susceptible bear to nuisance bear	m	0.001–0.030
Maximum catch probability	$\gamma(t)$	0.034–0.447 or 0–0.8
Carrying capacity	K	1099–1790
Strength of density effect	ω	$7–11 \times 10^{-4}$

Denoting the catch probabilities of susceptible and nuisance bears by $p_0(t)$ and $p_1(t)$, respectively, we have:

$$p_1(t) = \gamma(t)e^{-A(t)} \text{ and } p_0(t) = F\gamma(t)e^{-A(t)}, \tag{14.13}$$

where $\gamma(t)$ is the maximum catch probability in a year without acorn production; F is the false-catch coefficient quantifying the type I error. We assume that the catch probability increases with decreasing acorn production because the previously mentioned phenomenon of mass appearance occurs in years with low acorn production. We also assume that $\alpha(t)$ and $\gamma(t)$ depend on the management policy or the population status characterized by $N_0(t)$ and $N_1(t)$.

The catch in number is a random variable given by

$$Pr[C_i = X] = \binom{N_i}{X} p_i^X (1 - p_i)^{N_i - X}. \tag{14.14}$$

Table 14.1 lists the parameter values used in Ohta et al. (2012).

Ohta et al. (2012) took that the catch probability $\gamma(t)$ and the release rate $\alpha(t)$ change according to the following expressions:

$$\gamma(t+1) = \begin{cases} \gamma(t) - 0.1, \text{if } N_0(t) < W_{index} \\ \gamma(t) + 0.2, \text{if } N_0(t) \geq W_{index} \end{cases} \tag{14.15}$$

$$\alpha(t+1) = \begin{cases} \alpha(t) + 0.2, \text{if } N_0(t) + N_1(t) < N_{index} \\ \alpha(t) - 0.1, \text{if } N_0(t) + N_1(t) \geq N_{index} \end{cases}$$

Namely, the catch probability decreases (increases) if the number of susceptible bears is small (large), while the release rate increases (decreases) if the population size is small (large).

14.7 Performance of Adaptive Bear Management

Ohta et al. (2012) set two performance criteria; (1) the risk that the number of female adults drops below 250 once during the next century, and (2) the risk that the number of nuisance bears grows above 70 over 30 times during the next century. In recent years, the number of nuisance bears was often more than 200, with one-third of bears being females. Therefore, criterion (2) aims to halve the frequency of years with many nuisance bears. Criterion (1) aims to prevent genetic deterioration due to population decrease, because the genetic minimum viable population (MVP) size, the sum of adult males and females, is said to be 500. Ohta et al. (2012) called criteria 1 and 2 the conflict risk and the population collapse risk, respectively.

In order to compare the management performance between adaptive management and non-adaptive management, Ohta et al. (2012) considered four management scenarios, (1) both $\gamma(t)$ and $\alpha(t)$ are constant; (2) $\gamma(t)$ is doubled but constant and $\alpha(t)$ is variable; (3) $\gamma(t)$ is variable but $\alpha(t)$ is constant; and (4) both $\gamma(t)$ and $\alpha(t)$ are variable (Table 14.2).

In Scenario 1 with a constant catch rate that corresponds to the actual management for Oshima population in Hokkaido, the conflict risk exceeded 90% (Fig. 14.1). There was no population collapse risk, although model predictions represented in the 95% confidence interval showed a large magnitude of uncertainty. The current catch rate was too low to cause a population decline of less than 25% of the estimated population size in 2008.

Scenario 2 has less conflict risk, but this is due to the doubling of the catch rate, not the effect of varying the release rate. Scenario 2 also does not pose any extinction risks, but the conflict risks still remain high at 60%. By contrast, if the capture rate is variable under scenario 3, the conflict risk disappears but now the population collapse risk becomes close to 40%. Finally, the adaptive management scenario 4 can reduce both the conflict risk and the population collapse risk when N_{index} and W_{index} are properly selected (Fig. 14.2).

Table 14.2 Overview of scenarios with risks of management failure (%)

Scenarios	1	2	3	4
Ecological risk	0	0	39%	10%
Conflict risk	93%	60%	0	30%

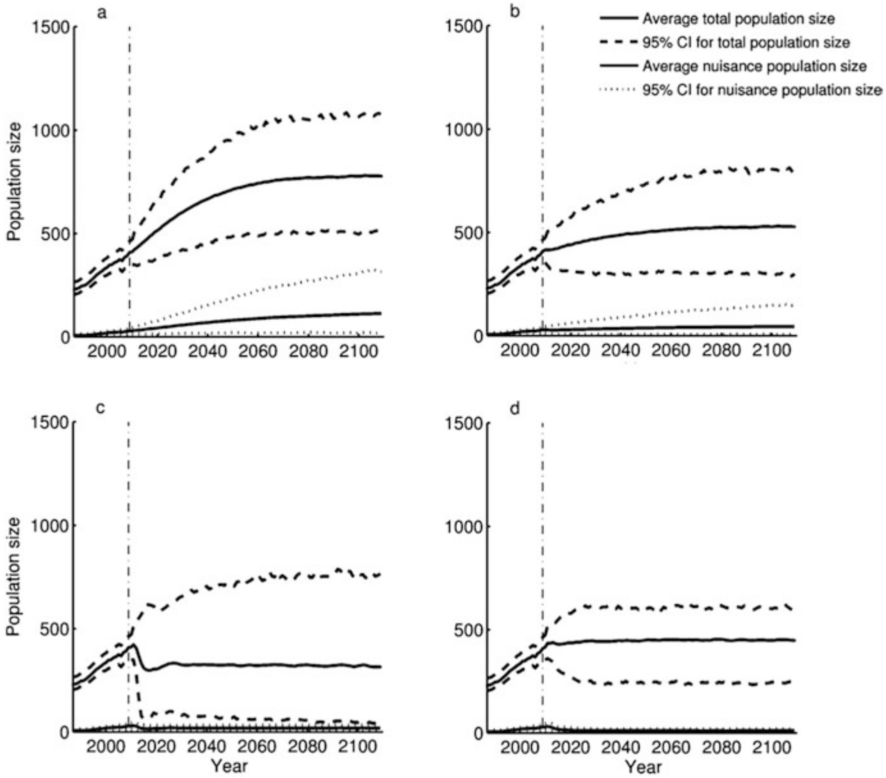

Fig. 14.1 The average number of total and nuisance female individuals with 95% confidence intervals from 1993 to 2110 under scenarios 1–4 (panels (**a**–**d**), respectively)

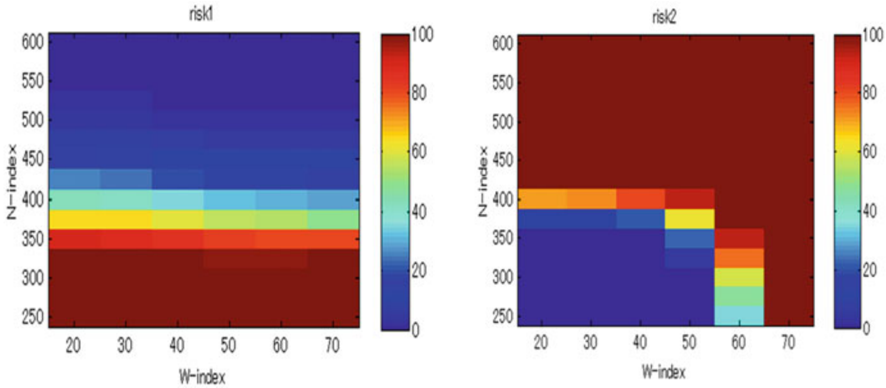

Fig. 14.2 Performance of bear management scenario 4 with variety of thresholds W_{index} and N_{index}. (**a**) conflict risk decreases with decreasing N_{index}, whereas the population collapse risk decreases with increasing both N_{index} and W_{index}. The optimal strategy is achieved when (W_{index}, N_{index}) = (10, 500)

14.8 Nuisance Bears Near the Shiretoko Natural World Heritage Site

Local stakeholders of Shiretoko National Park worry about photographers who take pictures of wild bears because bears accustomed to human presence may become a nuisance. Normally, wild bears avoid people, but after a while fear dissipates and bears may turn increasingly aggressive. Nuisance bears subsequently appear in urban areas and become targets for culling. The staff tasked with bear culling are neither soldiers of the Japan self-defense forces, nor police officers or national park rangers. They do not even belong to the Ministry of the Environment. Instead, the Shiretoko Foundation staff members and local game hunters take part in culling as needed.

Predicting emergency dispatches to deal with nuisance bears is impossible, and bears may appear around human settlements at any time, day or night. The number of cases handled in FY2012 and FY2015 exceeded 1000, which is far beyond the Shiretoko Foundation's capacity. In the Meeting of the Bear Working Group held on December 14, 2015, the response policy in the event of a personal injury was discussed, although such injuries have not yet been recorded in the Shiretoko World Heritage site (Fig. 14.3).

Bears usually avoid hunters with guns, yet these omnivorous animals are obviously stronger than people without guns. They often feed on deer, and if hungry, may perceive even humans as prey. It is most common, however, that bears who are no longer afraid of people rob lunch boxes or juice bottles carried by tourists and climbers. Humans and bears best coexist if they fear each other.

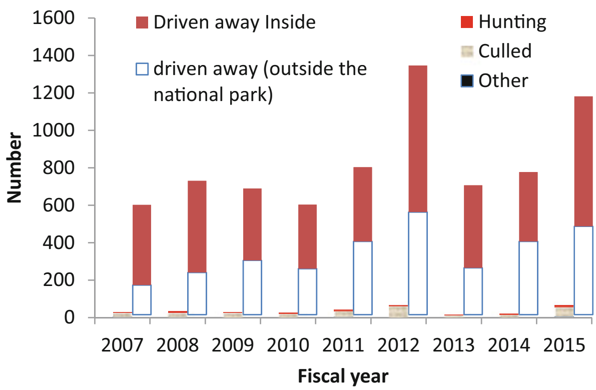

Fig. 14.3 Number of culled and driven-away bears in Shari-cho and Rausu-cho. Source: The Science Council for Shiretoko World Heritage, until November 2015

14.9 Driving Bears Away Is Ineffective

The Hokkaido Government released an estimate of its bear populations in 2015. According to the release, the number of nuisance bears is increasing more rapidly than the bear populations themselves. The declining number of hunters exacerbates the situation as they no longer need to fear people; hardly any encounter is lethal. At the same time, the area of the bear habitat has been spreading to the point that bears have appeared in the urban area of Sapporo city.

Bears who experience human food are dangerous. According to the Yellowstone National Park in the USA, "garbage kills bears," meaning that bears often develop a preference for food waste, which emboldens them to enter human settlements and attack people on occasion. Such bears become the prime targets for culling.

According to the "Ordinance on Biodiversity Conservation" formulated by Hokkaido Prefecture in 2013, feeding of wild animals is regulated throughout Hokkaido. Regulations are truly being enforced by the Hokkaido Government, which has taken measures to, for example, publicize names of violators (Fig. 14.4).

Feeding is not the only cause of bear habituation. Taking photos from a close distance also makes bears accustomed to people. The abovementioned ordinance does not regulate photography, which in October 2013 prompted the Shiretoko World Heritage Science Council to issue an urgent statement requesting photographers to keep distance from bears. The staff of the Ministry of the Environment and the Forestry Agency distributed leaflets with the statement to photographers on the Iwaobetsu River.

In the Bear Management Policy of Shiretoko Peninsula formulated in 2011, zones I and II, such as the World Heritage site, prioritize bears, whereas zones IV and V,

Fig. 14.4 Photographer within a close proximity to a bear. Photo by Shiretoko Foundation

such as farmland and urban areas, prioritize people. In zones I and II, the killing of bears is discouraged unless bears chase people and thus threaten human lives. Bears that often appear in photographs are marked as the targets to be driven away. It has been recognized, however, that even if the staff of the Shiretoko Foundation drive the same bear away dozens of times, the same bear appears again. The Shiretoko Foundation continues to persuade photographers, but unfortunately, the measure is proving to be ineffective.

Shiretoko is a relatively small home range for bears, who often wander off to the adjacent Shibetsu district. In order to keep people and bears separate, the town of Utoro has been surrounded by electric fences and bushes that cut off bear access to the town.

In conclusion, managing bears is necessarily accompanied with managing people. Coexistence cannot be achieved by considering solely human interests and values. Management measures must also be taken to prevent incidents and injuries arising from the human–bear conflict.

14.10 Bears Have Begun to Appear Even in Sapporo City

Since the end of the 2010s, brown bears often appear around the terminal subway stations in Sapporo, which is a city of almost two million (Fig. 14.5). The conflict between humans and wild animals has thus become more serious not only in national parks but also in metropolitan areas.

Population control, which rests on killing a limited number of individuals, is one of the measures for coping with the overabundant wild animals. For example, sika deer had a small population in the 1970s and were protected, causing the number of sika deer to increase considerably by 1998. The Sika Deer Conservation and Management Plan in Eastern Hokkaido was established in 1998 with the purpose of commencing a systematic population control. Hokkaido Prefecture was heavily criticized at the time for devising plans to cull deers, yet the control of deer populations at the World Heritage sites of Shiretoko and Yakushima (western Japan) has since been implemented with some success.

Unlike deers, bears cause damage that directly affects people's lives. Appearances of bears cause restrictions to human activities; for example, children cannot go to school, and sporting events, such as marathons, get canceled. Sapporo City furthermore urges its residents to "collect information about brown bear appearances from the web site by the municipality," stating also that "citizens should not approach places where bears have been spotted."

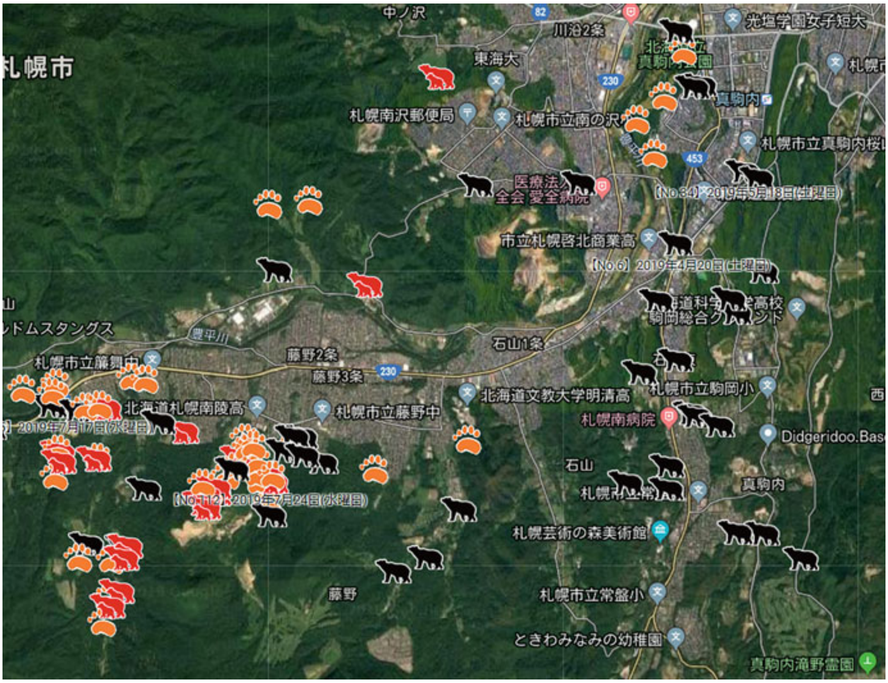

Fig. 14.5 Bear appearances in Sapporo City (http://www.city.sapporo.jp/kurashi/animal/choju/kuma/ by Google Map). Black labels signify sightings of solitary bears, red labels signify sightings of mother bears with children, and orange logos signify the sightings of bear footprints. Hunting is commonplace in urban areas adjacent to the forest, including near Makomanai Subway Station

14.11 Should We Exploit Wildlife?

Ban on hunting does not help protect wildlife, as evidenced by the examples of African elephants (*Loxodonta africana and L. cyclotis*) and black rhinoceros (*Diceros bicornis*) documented by the Convention on International Trade of Endangered Species of Wild Fauna and Flora (CITES). The CITES seeks to restrict international trade in endangered species. When agreed upon at the Conference of the Parties and listed in Appendix I, the international commerce of a target species is prohibited. Listing of a species in Appendix II requires permission from the exporting country.

African elephant ivory was traded at a high price, causing the population to decline from 1.34 million to 0.62 million between 1979 and 1989. Elephants were listed in CITES Appendix I in 1989. Populations in southern Africa (e.g., Zimbabwe) have since begun to grow, and have been downgraded to Appendix II. The export of ivory from southern Africa nonetheless remains scarce, with only two exceptions on record, in 1999 and 2009, called "one-off trades." Elephants are thus a successful example of protection measures by the CITES.

Black rhinoceros (*Diceros bicornis*) are sought after for their horns, which, similar to ivory, trade at a high price. The population was estimated at 65,000 in 1977, and despite listing in Appendix I of the CITES, the decline continued all the way to only 2410 individuals in 1995 (Emslie and Brooks 1999). Why could African elephants be protected from poaching, but black rhinos could not? Ivory has artificial substitutes, and the main markets were Japan, Europe, and the United States, where strict crackdowns could stifle the black market, ultimately preventing poaching. Black rhino, by contrast, is used as Chinese herbal medicine in Asia, and there is no substitute. It is, in fact, believed that poaching and smuggling have expanded since the ban on exports.

After a one-off trade of ivory from southern Africa in 2009, China's ivory market expanded, along with poaching and smuggling in eastern Africa, causing the elephant population in the eastern region to decrease. Poaching mortality was greatest around 2011 but has since declined (Kshatriya et al. 2019). Interestingly, Japan has stocks of ivory imported before the embargo in 1989, and there is no evidence that ivory from elephants poached in eastern Africa has ever ended up on the Japanese market.

It is not the case that humans exclusively threaten elephants. Dozens of people per year get mauled by elephants throughout Tanzania (Mduma et al. 2010). This type of human–elephant conflict is more critical around the nature reserves as shown by Mduma et al. (2010).

From the perspective of wildlife exploitation, wild animal meat (bushmeat), fish, shellfish, and even insects are important protein sources. This is especially true in poorer regions of the world, such as sub-Saharan Africa. According to the recommendations of the Convention on Biological Diversity held in 2011, "In the Congo Basin for example increasing population and trade from rural to urban areas compounded with the lack of any sizable domestic meat sector are the main causes of unsustainable levels of hunting[. . .] Therefore, there is no alternative to making the use of wildlife for food more sustainable" (CBD 2011). Preventing people, whose sustenance depends on wildlife, from hunting means cutting their source of dietary protein. The only way forward, at least in the near future, is then to manage exploitation in order to ensure the sustainable use of biological resources.

In developed countries, wildlife is sometimes considered a culinary delicacy, but hunting mainly occurs to avoid damage to agriculture, forestry, and fisheries, to protect natural vegetation, and to avert incidents and injuries. In Europe and the United States, where trophy hunting is thriving, this is a powerful means of managing animal-inflicted damage. Attempts to enjoy some positive aspects of game hunting have also been made in developing countries; for example, elephants and tigers are hunted as trophies, where the profits from such hunting are returned to the region. The IUCN Species Conservation Committee (IUCN/SSC 2012) also states that "[In some cases,] trophy hunting forms an important component of Community-Based Conservation and Community-Based Natural Resource Management, which aim to devolve responsibility for the sustainable use and management of wildlife resources from distant bureaucracies to more local levels."

Fig. 14.6 Schematic picture used in UNESCO's Man and the Biosphere (MAB) Program. Rather than isolating and preserving wildlife as shown on the left, wildlife should be treated as an open system, including people's settlements, as shown on the right

With changing dietary habits in the developed world, such as the advance of vegetarianism in Europe and North America, what constitutes acceptable food sources may change in the near future. Since the 1990s, for example, there has been a movement to avoid canned tuna caught by the fisheries with substantial dolphin bycatch. Crops grown on farmland devoid of wild birds and beasts may also become unacceptable. Farmland has historically been inhabited by wild birds or beasts, but they had been driven out. If farming is to coexists with wild birds and beasts, the human–wildlife conflict may become inevitable in agriculture, forestry, and fisheries. People are inseparable from the biosphere (Fig. 14.6), and thus not only use wildlife but are sometimes being used by wildlife too.

There are cases in Japan of wild bears preying on people. Crops and seafood grown for human consumption are sometimes robbed by wildlife, deers, bears, boars, monkeys, etc. Capturing and using wildlife is one of the measures to protect people's lives from harm. At the same time, animals whose encounters with people result in negative experiences are more likely to avoid such encounters in the future. It can thus be said that humans and wildlife do not make friends but can coexist by respectfully fearing each other.

The population of developing countries is still growing, and as a consequence, the conflict between humans and wildlife is on the rise. Economic development further exacerbates the conflict. Once a developing country has progressed to the developed status, land use is usually at a point where preserving primeval nature is possible only within a system of national parks or similar reserves. There is, however, a limit to coexisting in a manner that separates the human sphere from the wildlife's biosphere. Intense conflicts always occur at the boundary, and to reduce such

Fig. 14.7 Electric fence to prevent bears invasion at Shari Town. Photo by Shiretoko Foundation (https://www.shiretoko.or.jp/higuma)

conflicts, it is necessary to open up substantial buffer zones. Fencing off human settlements to prevent the intrusions by wildlife is difficult unless either the nature reserve or the settlement is small enough. An example of this in Shiretoko is the Utoro town area which has been fenced off to stop or, at least, reduce brown bear sightings where people live (Fig. 14.7).

Wildlife exploitation is also a concern from an epidemiological point of view. A new type of coronavirus disease (COVID-19), which turned into a global pandemic in March 2020, is believed to be of zoonotic origin. As long as people use wildlife for food, the risk of contracting viral or parasitic diseases cannot be ruled out. When zoonotic infections begin to transmit from person to person, it is unrealistic to expect an epidemic that will remain contained in developing countries. The disease can explode worldwide within a few months, and turn into a global pandemic as was the case with COVID-19.

It is important to recognize that the perception of the human–wildlife conflict has changed over the course of time, and depends on the way of life. A quarter of the century ago, Japan was free of any serious human–wildlife conflict. Many wild birds and beasts were recognized as threatened and protected accordingly. Successful protection eventually replenished wildlife populations, and nowadays Japan and other developed countries have to live with incidents whereby, for example, bears invade cities and injure people. Japan's rich nature and typically mountainous terrain have helped local wildlife to bounce back.

In terms of the way of life, urban dwellers, unlike their rural counterparts, rarely suffer in a direct manner from harm caused by wildlife. This causes the divergence of views on nature conservation and animal welfare. Ultimately, neither side is "right" or "wrong." It is instead crucial to understand the underlying reasons for the divergence of views. Behavioral experiments could be particularly helpful in this context.

Human society cannot afford to simply block the consumption of wild animals. In Japan, black bears in the Shikoku Island, whose population comprises a few dozen individuals, is not targeted for killing. However, it is not always the case that a threatened species is really at high risk of extinction. Moving to prohibit all killings in such cases could substantially narrow the spectrum of available measures aimed at reducing the human–wildlife conflict.

Under the Convention on Biological Diversity, the basis of nature conservation is that there are ecosystem services or nature's contribution to people that promote human well-being. This leads to the idea of Sustainable Development Goals, according to which the conservation of nature is not a goal per se. Instead, conservation is an indispensable measure for advancing the well-being of human populations. Furthermore, conservation and utilization are not necessarily irreconcilable. In some parts of the world, in fact, there is no substitute for the latter. Using wildlife products is thus one of the major ecosystem services.

The problem of animal-inflicted damage in developed countries such as Japan and the problem of conservation and utilization of wildlife in developing countries should be resolved in coordination with local residents. Doing so, unfortunately, has not yet become an established practice. The prevailing thinking is likely to change in the near future, probably due to its infeasibility. We are facing many open questions.

The CITES is a convention that bans or controls international trade but does not address domestic consumption. Because the European Union (EU) prioritizes free trade within the EU, trade in wildlife products is possible between the EU countries even if this is prohibited by the CITES. There is also a potential conflict of interest in that CITES does not list endangered species in appendices automatically upon obtaining sufficient scientific evidence that a species needs protection. The member states must agree first. In sum, current practices in wildlife management have much room for improvement.

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Part IV
Ecosystem-Based Risk Management

Chapter 15
Effects of Dams on Ecological Risk of Inland Fishes

Hiroyuki Matsuda and Kentaro Morita

Abstract Dams are effective to prevent flood disasters and effectively use water resources. However, the impact on the ecosystem is concerned, such as the dam blocking the movement of organisms upstream and downstream. Moreover, it is not possible to prevent a disaster once every several 100 years. Originally, it was designed with a balance between disaster risk and ecological risk, and these risks are not completely eliminated. Which risk is to be emphasized changes from decade to decade, and it should be different, for example, between nature reserves and the rivers flowing through large cities. In this chapter, using a generalized linear model, we propose a regression equation that predicts the survival probability of the char population upstream of the dam from the years after the installation of the dam and the catchment area. Using this model, we extrapolated the future extinction risk of local populations. In addition, population viability analysis is performed using life table analysis to predict extinction risk. At that time, we considered the annual fluctuation of the subscription rate. In this way, both statistical analysis based on distribution information and population modelling based on life history parameters suggest that the risk of extinciton increases due to the division of habitat by dams.

Keywords Salmon · Population viability analysis · Life table analysis · Flood control · Generalized linear model

15.1 Background

It is said that a Japanese warlord, Nobunaga Oda, in the sixteenth century said, “A person who governs water governs the country.” Flood control was a long-standing national policy of Japanese, which has many mountains and short rivers. After the

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second world war, many dams were created in Japan in order to prevent floods, secure agricultural water, and generate hydroelectric power.

The dam has been criticized as an example of bad public works. Some dams have been removed in the United States. Also in Japan, the Arase Dam in Kumamoto Prefecture was removed in 2016. The biggest reason for criticizing a dam is that it is damaging natural ecosystems (Hart et al. 2002).

Why do dams that prevent floods and keep the amount of water constant destroy nature? We will explain the reason in this section. It is also a good exercise that evaluates the extinction risk of local populations.

Why is dam damage to the ecosystem? First, (1) diadromous fish such as salmon and trout, that migrate between the sea and the freshwater, are the nature most affected by dams. Many salmon and trout spawn in rivers, descend into the sea, grow, and return to the river. If there is a dam, it will not be possible to ascend from the downstream side to the upstream side. (2) The amount of water flowing downstream, the amount of sediment, and the amount of organic substances change, which affects the downstream ecosystem as explained below. (3) There will be no floods or changes in water volume necessary to maintain downstream ecosystems. Furthermore, (4) When a dam is constructed, a dam lake will be created, and the place of habitation of the birds of prey that originally lived there will be lost. In Japan, point (4) was especially emphasized, and the dam construction plan sometimes stopped when a nest such as bear hawks was found at the site. However, at the Tokuyama Dam in Gifu Prefecture, which is the largest dam, it was predicted that the home range of four bear hawks would be greatly degraded. However, after the dam was put into service, the raptors still lived in the same place (Water Resources Organization Tokuyama Dam Management Office 2013).

Explain the impact on the downstream, especially the negative impact of stopping the flood. A Greek historian Herodotus described it as “The Egyptian gift of the Nile.” The river was flooded and became fertile and could be used for agriculture. Instead of building a dam to prevent floods, the ancient Egyptians developed solar calendar-based astronomy and predicted the flood season. Biodiversity is maintained by a heterogeneous mosaic of transitions and natural disturbances (Christensen et al. 1996). Disturbances include forest fires, landslides, fallen trees due to typhoons, floods, etc. A flood sometimes changes the flow path (thalweg) and develop braided channels and floodplain. Some plants grow in the early stages of succession and grow in an empty land caused by flooding. Salmon redds are often constructed on disturbance-mediated river bed. A monocarpic perennial species, *Aster kantoensis* distributes on the Kinugawa and Tama rivers, but it is endangered because of lack of flooding (Washitani et al. 1997) (Fig. 15.1).

Some dams have begun to attempt artificial emissions to disturb the downstream ecosystem. The US Glen Canyon Dam, which was constructed on the Colorado River in the 1960s, was the precursor, as introduced below. The Colorado River used to be characterized by severe fluctuations, and the annual water level change of Lake Powell created by the dam reached 60 m. After the dam was built, the amount of water downstream became stable and hydroelectric power was generated. However, the dam has transformed the ecosystem of the Grand Canyon National Park

Fig. 15.1 Views of the Danube (front) and the new Danube flowing through Vienna from the Vienna Forest, Leopoldberg, Austria. There is a dam for hydroelectric power generation slightly downstream from this photograph. It used to meander in the past, but it is becoming more straight (photo by H.M)

downstream of the dam. Previously, annual plants mainly grew on the waterside, but dams have prevented floods and the number of perennial plants including exotic species has increased. The sediment (sand, gravel) carried by the upstream has decreased and the sandbar has stopped growing, the pool where fish lived disappeared. The gravels needed for salmon spawning beds and substrate for various aquatic species have disappeared.

After that, it was recognized that seasonal floods were an important factor for sustaining a healthy ecosystem in the Grand Canyon, and the Glen Canyon Conservation Law was enacted in 1992. Controlled release of water is necessary to revive the downstream ecosystem, and release is controlled under adaptive management (the release plan is reviewed while continuing the monitoring, which will be described later).

Apart from the viewpoint of nature conservation, there are criticisms that dams do not always perform the functions that they should have. (5) Other flood control projects such as levees and river bed excavations can also function sufficiently at a lower cost than dams. (6) Dams often take away the space needed for ethnic minorities and culture. There is an example of Nibutani dam in Hokkaido in Japan. (7) Some dams were planned to create effective demand apart from the true need. The Hoover Dam, built on the Colorado River, is a famous example of a New

Deal policy that also appears in history textbooks. However, the reason why dams are currently being reviewed in the United States is due to nature conservation rather than such social and economic criticism (Hart et al. 2002).

It is widely recognized that dam flood control policies rather damage the ecosystem. One of the biggest problems is the blockage of the movement route of fish, etc., by the dam and the division of the habitat. Fish downstream of the dam will not be able to ascend upstream, reducing the environmental carrying capacity of each population and increasing extinction risk. In addition, the reduction of survival rate, maturity rate, and reproductive rate due to impaired migration also increases the risk of extinction. A method for assessing the extinction risk of a local population based on basic information on the population size and structure, survival rate, and reproductive rate is called population viability analysis (PVA). The individual-based model (IBM) is often used to calculate the extinction risk.

15.2 Extinction of Freshwater Fish

We apply the regression model to the relationship between the year of dam in-service and the disappearance of the local population of fish. Salmonid fishes migrate between the freshwater and the sea, but not all. Many chars spend their lives on the river. However, some descend into the ocean, returning to the river, much larger than a normal char.

Although no direct evidence of a causal relationship has been shown, there are results of investigating the extinction status of chars in the upstream part of 52 dams in Hokkaido (Morita and Yamamoto 2002). When a dam is created, the upstream region is isolated, so the longer the isolation periods (years, t) and the narrower the habitat (basin area w), the more likely the char will be extinct. These trends are clearly visible in Fig. 15.2. By the way, all 32 populations were inhabited without dams (Morita and Yamamoto 2002).

Let us consider the relationship between dams and occurrence of chars using a statistical method called a logistic regression model (Dobson and Barnett 2018). This is because when there are data such as the number of years since the construction of a certain dam and the area of the basin, the probability p that the char is extinct in the upstream region of the dam is

$$\frac{p_i}{1-p_i} = \exp[a + b \log t_i + c \log w_i + d \log s_i] \tag{15.1}$$

where a, b, c, d mean the proportional constants; t_i, w_i, s_i represent the number of years after the dam was made, the area of catchment, and the gradient of river for dam i, respectively, and other explanatory variables are used (Morita and Yamamoto 2002).

We obtain the best fit model that maximizes the following logarithmic likelihood L (Dobson and Barnett 2018):

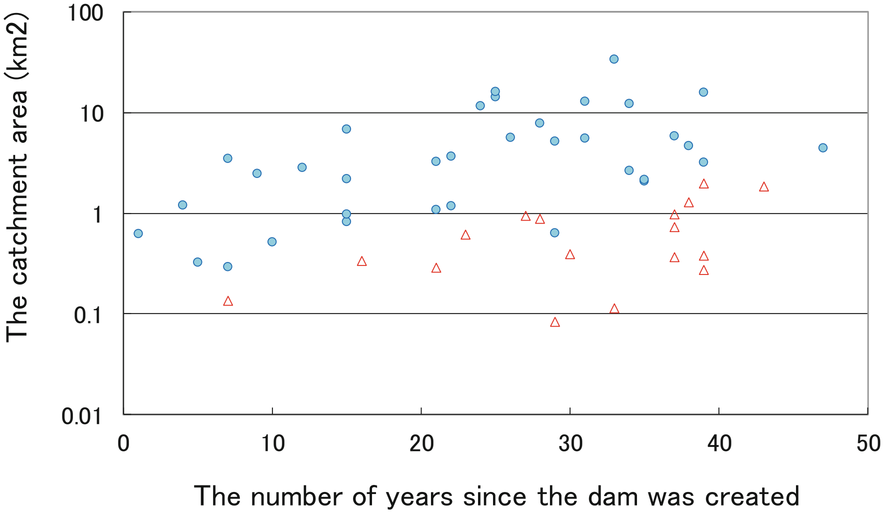

Fig. 15.2 Relationship between the number of years since the dams were created in 52 dam upstream areas of Hokkaido and their habitats were isolated, the catchment area in the upstream area, and the habitat status of char. Triangles indicate a river where char is extinct, and circles indicate a river where it inhabits (modified from Morita and Yamamoto 2002)

$$L = \sum_{i=1}^{n} [q_i \log p_i + (1 - q_i) \log (1 - p_i)] \tag{15.2}$$

where p_i is the probability that char exists in dam i, x_i is 1 or 0 if char exist or does not exist in dam i, respectively. We obtained the set of parameter values a, b, c, d that maximize the logarithmic likelihood.

$$\frac{p_i}{1 - p_i} = \mathrm{Exp}\,[7.02 + 2.32 \log t_i + 1.74 \log w_i + 0.1 \log s_i] \tag{15.3}$$

The regression equation was calculated. The logarithm is the natural logarithm. Using the obtained regression equation, the results show that the predicted p is less than 50% but still exist in four dams, and the predicted p is still more than 50% but have already disappeared at five dams.

15.3 Prediction of the Future Extinction Risk from the Regression Model

If the basin area and slope will not change in the future, if only the isolation time is increased, t in the regression Eq. (15.4) can be replaced with $t + \Delta t$, and the extinction probability p_i at each dam i after Δt years can be calculated. As shown in Fig. 15.3, the prediction is quite pessimistic. Since the extinction probability of each river can be predicted, this regression equation can be verified by a follow-up survey. Extinction occurred in three sites in a 15-year follow-up survey (Morita et al. 2019).

There are some unexplained dams in the regression model (15.3), as you can see by looking at year 0 (present) in Fig. 15.3. It may be helpful to consider the reason for each exception to get a better regression model. Risk assessment should not be something that can be calculated. It is always necessary to examine the reality of the results and how they can be verified in the future, even if it is impossible at the present. What is proven at this point alone cannot address environmental issues, but evaluation that cannot be permanently verified is not science.

By the way, chars seem to be vulnerable to acidification. In Norway, acidification has had a serious impact on freshwater fish since 1960, of which 190 out of 5666 char populations have become extinct due to acidification (Hesthagen and Sandlund 1995). So far, the effects of acid rain on freshwater fish are not known in Japan, but we are worried about the future.

15.4 The Life-Table Analysis of Char

In order to calculate the risk of extinction, it is necessary to estimate how many newborn eggs will hatch, grow, and mature. Some fish, including chars, take several years to reach maturity and continue to spawn alive the year after spawning (i.e.,

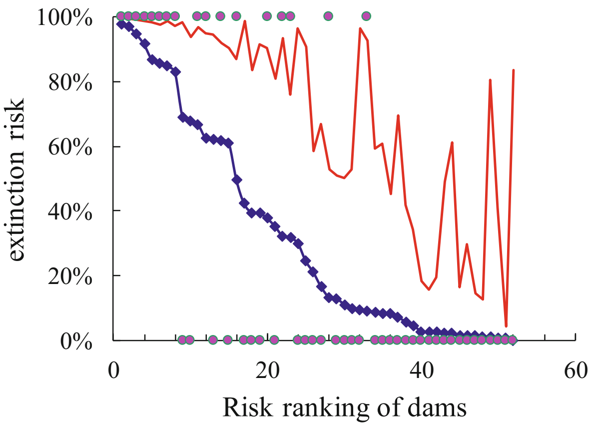

Fig. 15.3 Extinction risk at present (◆ mark) and 50 years later (red line) regarding the survival of char in the 52 upstream dam areas shown in Fig. 1. The dam with a ● in 100% indicates that the char is gone, and the mark with 0% indicates that it is still alive

iteroparous). As shown in Table 15.1, the survival rate, maturity rate, and fecundity of Japanese char are relatively well investigated.

The average generation time from birth to spawning also influences the population trends. The percentage of individuals who survive from birth to age *a* is called the survival rate at age *a*, l_a. The survival rate per year (annual survival rate) from the spawning season of *a*-year-old to the spawning season of the following year can be represented by the survival rate ratio l_{a+1}/l_a. The initial survival rate (including hatching rate) from 0 to 1 year old is low, and the yearly fluctuation due to environmental stochasticity is severe.

There are individual differences in growth, and some mature at the age of 2 and some do not mature at the age of 4. The ratio of mature individuals among all *a*-year-old individuals is called the *a*-year-old maturity rate. As shown in Table 15.1, this is different between males and females. Many fish continue to grow after maturity, and the number of eggs laid continues to increase with age. However, since the growth slows down after the age of 7, the number of eggs laying over the age of 7 is assumed to be constant here.

The initial survival rate of 0.035 is the geometric mean of the annual survival rates of the two species in Table 15.2. After that, the survival rate up to the age of *a* is given by $0.035 \times 0.4^{a-1}$, assuming an annual survival rate of 0.4. The sex ratio at birth is 1:1 so half of the total laying eggs will be female. In the char, the spawning season arrives about 2 months before the birthday, and the number of individuals in the spawning season is measured. Therefore, the term 1-year old is sometimes written as "0+" years old. $l_aM_aE_a/2$ represents the contribution to reproduction at each age, but it is clear that breeding at ages 3 and 4 is the main activity. This sum $\Sigma_a l_aM_aE_a/2$ represents the net population growth rate. The table only shows up to

Table 15.1 Life history parameters of char (Morita and Yokota 2002)

Age a	Annual survival rate S_a	Survivorship ℓ_a	Male maturation rate	Female maturation rate M_a	The number of spawned eggs E_a	$\ell_aM_aE_a/2$
0	0.035	1	—	—	—	—
1	0.4	0.0350	0	0	0	0.0000
2	0.4	0.0140	0.84	0.09	129	0.0829
3	0.4	0.0056	1	0.84	201	0.4721
4	0.4	0.0022	1	0.96	264	0.2849
5	0.4	0.0009	1	1	307	0.1375
6	0.4	0.0004	1	1	330	0.0591
7+	...		1	1	330	...

Table 15.2 Yearly change in the initial survival rate of char (Morita and Yokota 2002). It also includes data on related species. The author name after the species name indicates the source

S. leucomaenis—Miura (1977)						*S. malma*—Kitano (1997)	
1965	1966	1967	1968	1969	1970	1992	1993
0.03	0.022	0.159	0.031	0.048	0.024	0.023	0.026

6-years old, but since it continues to live after 7-years old, after 6-years old it is a geometric sequence with the first term of 0.0591 and a common ratio of 0.4. $\Sigma_a l_a M_a E_a/2 = 0.9774 + l_6 M_6 E_6/2/(1\text{–}0.4) = 1.076$. When this is greater than 1, the population continues to grow. However, when the initial survival rate fluctuates year by year, it is not always appropriate to take the geometric mean as described later. The fate of the population depends not only on the average and variance of the life history coefficients but also on whether their high years continue (see the chapter on bears).

The method shown in Table 15.1 can also be expressed using a matrix. This is called the Leslie matrix if all individuals die before 7-years old (Caswell 1990). For ages 7 years and older, the maturity rate, annual survival rate, and egg production were all considered to be the same as those at age 6, so the numbers of individuals aged 6 years and older are collectively shown. The formula for the population varies according to the age composition matrix corresponding to Table 15.1

$$\mathbf{N}_{t+1} = \mathbf{L}.\ \mathbf{N}_t \tag{15.4}$$

or

$$N_{t+1,a} = \sum_{j=1}^{6} L_{a,j} N_{t,j}$$

where column vector $\mathbf{N}_t = (\boldsymbol{N}_{t,\ 1}, \boldsymbol{N}_{t,\ 2}, \ldots, \boldsymbol{N}_{t,\ 6})^{\mathbf{T}}$ consists of the number of female individuals of age i at year t, denoted by $N_{t,i}$; matrix $\mathbf{L}$ consists of L_{ij}, where

$$L_{1,a} = \frac{S_0 M_a E_a}{2} \text{ for } i = 1$$

$$L_{a+1,a} = S_a$$

$$L_{i,j} = 0 \text{ for other } i, j$$

The components of $\mathbf{L}$ are the number of females individuals of age a at year t. Since the number of female individuals is important for the viability of the population, we ignore the number of male individuals.

For the numerical values in Table 15.1, the matrix calculation is as follows.

$$\begin{pmatrix} N_{1,t+1} \\ N_{2,t+1} \\ N_{3,t+1} \\ N_{4,t+1} \\ N_{5,t+1} \\ N_{6,t+1} \end{pmatrix} = \begin{pmatrix} 0 & 0.207 & 2.949 & 4.451 & 5.369 & 5.772 \\ 0.4 & 0 & 0 & 0 & 0 & 0 \\ 0 & 0.4 & 0 & 0 & 0 & 0 \\ 0 & 0 & 0.4 & 0 & 0 & 0 \\ 0 & 0 & 0 & 0.4 & 0 & 0 \\ 0 & 0 & 0 & 0 & 0.4 & 0.4 \end{pmatrix} \begin{pmatrix} N_{1,t} \\ N_{2,t} \\ N_{3,t} \\ N_{4,t} \\ N_{5,t} \\ N_{6,t} \end{pmatrix} \tag{15.5}$$

Do not make a mistake as follows. Please note that this mistake is frequently done even by experts.

$$\begin{pmatrix} N_{0,t+1} \\ N_{1,t+1} \\ N_{2,t+1} \\ N_{3,t+1} \\ N_{4,t+1} \\ N_{5,t+1} \\ N_{6,t+1} \end{pmatrix} = \begin{pmatrix} 0 & 0 & 5.9 & 84.3 & 127.2 & 153.5 & 165 \\ 0.035 & 0 & 0 & 0 & 0 & 0 & 0 \\ 0 & 0.4 & 0 & 0 & 0 & 0 & 0 \\ 0 & 0 & 0.4 & 0 & 0 & 0 & 0 \\ 0 & 0 & 0 & 0.4 & 0 & 0 & 0 \\ 0 & 0 & 0 & 0 & 0.4 & 0 & 0 \\ 0 & 0 & 0 & 0 & 0 & 0.4 & 0.4 \end{pmatrix} \times \begin{pmatrix} N_{1,\tau} \\ N_{2,\tau} \\ N_{3,\tau} \\ N_{4,\tau} \\ N_{5,\tau} \\ N_{6,\tau} \end{pmatrix} \tag{15.6}$$

When calculated according to Eq. (15.6), a 3-year-old parent lays 201 eggs at year t. It exists as an egg at year $t + 1$, and at year $t + 2$ about 7 individuals survive to age 1 with an initial survival rate of 3.5%. In fact, a parent who is 3 years old at t year should lay 201 eggs in that year, and about 7 individuals should survive to 1 year old at $t + 1$ year. So Eq. (15.6) is incorrect.

If the population size and its age composition in the next year can be calculated using the same age composition matrix every year, the age composition will eventually converge to stable age distribution and continue to increase at a constant population growth rate. It is given by the maximum eigenvalue of the above transition matrix and the corresponding right eigenvector.

The transition matrix based on age-structured population is called the Leslie matrix in ecology. However, in the above, $N_{6,t}$ is grouped together for ages 6 and older and simplified. Note that since the survival rate after age 7 is low, ignoring it will result in the maximum eigenvalue λ of 1.01, which will underestimate the natural rate of increase to approximately half. Thus, when discussing the rate of increase or decrease of 1% per year, it cannot be ignored that there is 1% of elder aged fish. Also, the eggs of older fish are larger than the eggs of younger fish and may be more effective than the number of eggs laid (Barneche et al. 2018). Furthermore, when the initial survival rate fluctuates, it is known that increasing the number of reproductive cycles contributes to the survival of the population even if the number of eggs laid throughout the life is the same (Katsukawa et al. 2002).

15.4.1 Population Dynamics Considering Process Errors

The previous section assumed that the survival rate and reproductive rate were constant every year. However, the actual number of individuals depends on (1) stochastic processes in survival and reproduction, and (2) the expected survival rate and reproductive rate themselves change depending on the environmental conditions (environmental stochasticity).

Survival is a stochastic event, and if the survival rate is 40%, more than or less than 4 individuals among 10 individuals survive in the next year even if the annual survival rate is 0.4. The probability of surviving m individuals at the next year with the annual survival rate S, denoted by $\Delta\mathrm{B}[m, m, S]$, is expressed by the following cumulative probability (when $m = 0$).

$$\mathrm{B}[n, m, S] \equiv \sum_{m'=0}^{m} \binom{n}{m'} S^{m'} (1 - S)^{n-m'} \tag{15.7}$$

where

$$\Delta\mathrm{B}[n, m, S] = \mathrm{B}[n, m + 1, S] - \mathrm{B}[n, m, S]$$

We can describe the following stochastic equation

$$\Pr[N_{a+1,t+1} = x] = \mathrm{B}\left[N_{a,t}, x, S_a \mathrm{Min}\left(\frac{K}{N_t}, 1\right)\right] \tag{15.8}$$

$$\Pr[N_{1,t+1} = x] = \mathrm{B}\left[\sum_{a=2}^{6} \frac{N_{a,t} M_a E_a}{2}, x, S_0\right]$$

Here we consider the density effect and assumed that when the number of adult fish $N_t = \sum_{a=2}^{6} N_{a,t}$ exceeds the carrying capacity K, the survival rate decreases in proportion to K/N_t.

If the annual survival rate $S = 0.4$ and the number of females this year is $n = 100$, the upper and lower limits of the 95% confidence interval, i.e., $0.025 < \mathrm{B}[n,m, S] < 0.975$, for the number of females in the next year are 31–50. Therefore, there is almost no fear of a sudden decrease. However, if $n = 10$, there are 1–7 individuals,

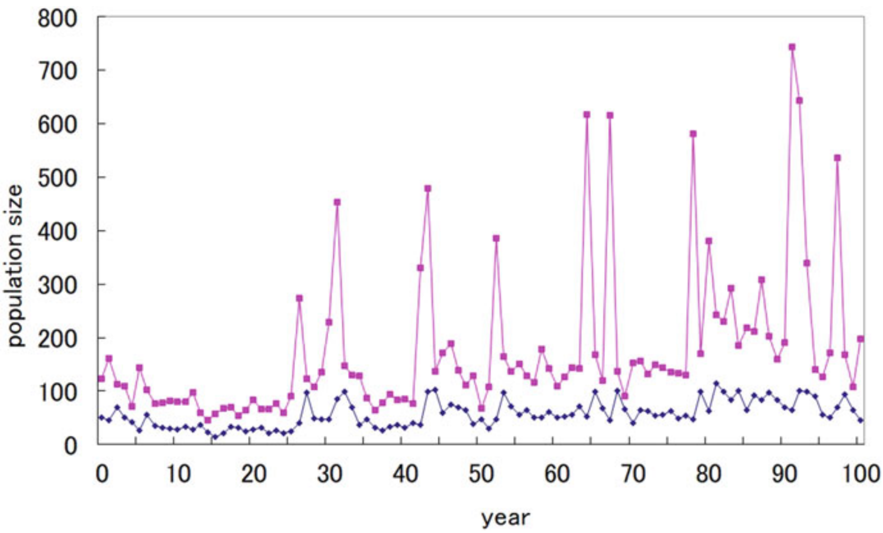

Fig. 15.4 An example of population variation using the life tables in Tables 15.1 and 15.2. Environmental change probability, population change probability, and density regulation (carrying capacity is 100 adult individuals) are considered. Hull squares and closed circles indicate the total number of individuals and the number of individuals aged 2 years or older, respectively (modified from Morita and Yokota 2002)

and there is a danger of a population collapse. The effect of stochastic fluctuations that occurs when the number of individuals is small is called demographic stochasticity.

Regarding the environmental stochasticity, especially the reproductive rate and initial survival rate are often affected by environmental conditions in comparison with the adult survival rate. In the case of char, as shown in Table 15.2, the initial survival rate varies greatly depending on the year.

As calculated using the age matrix model, the average number of chars should increase. However, demographic and environmental stochasticity can lead to unlucky reductions. There is a risk of extinction.

Regarding environmental stochasticity, the reproductive rate and the initial survival rate are particularly susceptible to environmental conditions. In the case of char, as shown in Table 15.2, the initial survival rate varies greatly depending on the year.

As calculated using the age matrix model in (15.6), the average number of chars should increase. However, demographic and environmental probabilities can lead to unlucky reductions. There is a risk of extinction.

Furthermore, we assume that the initial survival rate S_0 fluctuates annually as shown in Table 15.2. Longer term observations are needed, but here it is assumed that the eight numbers shown in Table 15.2 appear with equal probability every year. This allows for environmental fluctuations.

Figure 15.4 shows an example of population variation obtained by performing a computer experiment on the abovementioned stochastic age-group population dynamic model. The population growth rate using the geometric mean of the initial survival rate increased by 2% per year, but it does not always increase. When the recalculation is repeated, different results are obtained each time because random numbers are used, and it may become extinct within 100 years as in this example.

When the initial survival rate varies year by year in an age structure model with overlapping generations, the long-term population growth rate even matches with the age structure model using the average initial survival rate or the geometric mean value (see chapter for bear management).

Assuming that the environmental capacity of individuals over the age of two is 50–250, and the initial number of individuals is half the environmental capacity. At this time, there are only a few females over the age of three. The demographic probability is so large that it cannot be ignored. The initial survival rate is once every 8 years, five times higher than normal years, and the rest is between 0.022 and 0.048. The number of individuals is gradually decreasing, and a situation in which a large number of individuals occasionally join is also shown in Fig. 15.4.

There is a nearly 2% chance that a "mast year" once every 8 years will not last for 30 years. In the meantime, if the annual average population declines by 3.7%, the population will decrease to 1/3 in 30 years. The probability that such a "30 years of the ice age " will be included somewhere within the 100 years is more than 70%. It may become extinct if there are no female or sire or years without membership. In Fig. 15.4, after the number of individuals decreased sharply, a new year was born in the 76th year, but the population could not recover and became extinct.

As shown in Fig. 15.4, the population size fluctuates considerably. We started here with half the number of fish in the carrying capacity, but it can sometimes exceed the carrying capacity. Even if the population growth rate in the matrix in (15.5) is 2%, it does not exceed beyond the carrying capacity. Even if 300 eggs are born, the survival rate up to the age of 2 is 1.4%, so there is a 2% chance of stock collapse, according to the binomial distribution. If there are only a few parents, there is a risk of collapse, and the probability of extinction sometime within 100 years is very high, as shown in Fig. 15.4.

15.5 How to Assess the Impact of Dam Construction on Wild Fish Populations

What will change in the simulation results shown in Fig. 15.4 if we build a dam? First, we consider that the habitat is fragmented and the environmental carrying capacity K is reduced. Figure 15.5 shows the risk of extinction within 100 years after 100 simulation experiments. When K = 100, the extinction risk after 100 years is about 15%, but when $K = 50$, the risk increased to about 70% (Table 15.3). However, if the population is divided and there are two populations, the risk that both populations go extinct is about the square of 70% or about 50%. Even so, the risk of extinction is lower in the case that the combined population with $K = 100$.

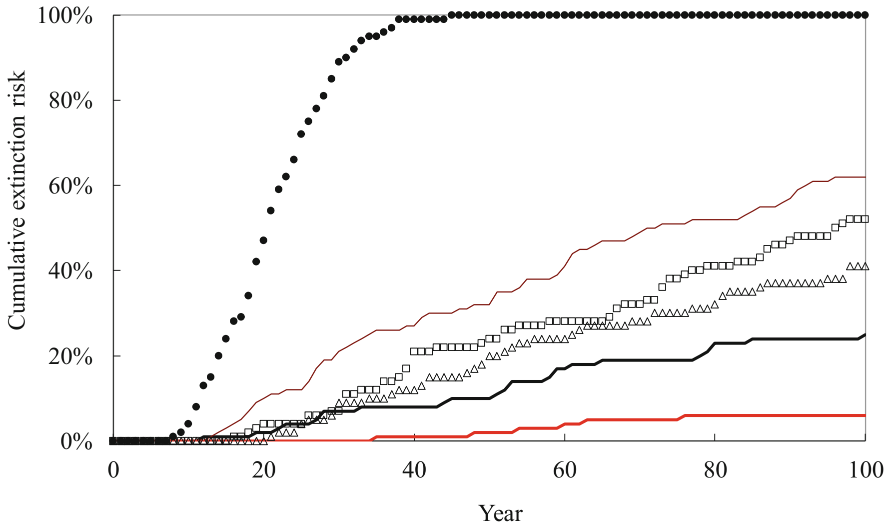

Fig. 15.5 Extinction risk within the next 100 years obtained by 100 numerical simulations. When the carrying capacity, K, is changed from 100 (thick black line) to 50 (thin brown line) or 200 (thick red line), either the initial survival rate (open triangles) or the annual maturity rate after 1-year old (open squares) is reduced by 5%, the case of halving (reduced by 50%) the initial survival rate (black circles) are shown (modified from Matsuda and Morita 2003)

The risk of extinction by year t, denoted by $p(t)$, is a function of year t, and the integral $\int_0^\infty [1 - p(t)]dt$ represents the average waiting time until extinction (corresponding to life expectancy in health risk), and the extinction risk is constant every year (where it is expressed by an exponential function such as $p = 1 - e^{-at}$, and the above-integrated value becomes $1/a$), its reciprocal represents the extinction risk per year.

The minimum number of individuals required to sufficiently reduce the extinction risk is called the minimum viable population (MVP = Minimum Variable Population) size. We often define MVP as the population whose extinction risk within the 100 years is less than 5%. It is notable that the risk is not zero. MVP is used as a measure of the urgency of protection measures. This risk is still much higher than the extinction risk of "natural conditions" without human influence. According to Table 15.3, the population whose $K = 200$ is not sufficient for the MVP. However, if the mean value or variance (or autocorrelation) of survival rate and initial survival rate are slightly changed, the quantitative conclusion changes and the value used is not sufficiently accurate. Also, the values estimated in one area in a species are not the same in other places or in related species.

Carrying capacity of the fish population is not solely affected by dams. Freshwater fish are not always in the same place throughout life, and many larvae and adult fish migrate from the birthplace. In that case, the juveniles that went downstream cannot run up the dam, and the survival rate would decrease for the upstream population in comparison with a case where fish that went downstream come

Table 15.3 The extinction risk and population growth rate for a variety of the carrying capacity, adult survival rate, initial survival rate, and maturity rate

Carrying capacity K	100	50	250	200	100	100
Annual survival rate of adults S	0.4	0.4	0.4	0.4	0.38	0.4
Initial survival rate S_0 factor	1	1	1	1	1	1
Maturation rate M_a factor	1	1	1	1	1	0.95
Extinction risk after 30 years	2%	13%	0%	0%	7%	1%
Extinction risk after 100 years	15%	72%	0%	10%	64%	33%
Geometric mean of the maximum eigenvalues of **L**[a]	6.1%				2.2%	4.6%
Geometric mean of population growth rate[b]	2.0%				−1.8%	0.6%

[a, b]See the text for detail

back. As is often the case with salmonids, some individuals of chars go to the sea. There is dimorphism in white-spotted char, resident form, and migrant form. The survival rate of the migrant form will be low, but when it returns to the river, it will be much bigger than the resident form that stays in the river. It is the same species as char, the resident and migrant forms are called "Iwana" and "Amemasu" in Japanese, respectively. The degree of anadromy in Hokkaido is particularly high. In rainbow trout (*Oncorhynchus mykiss*), it is also called rainbow and steelhead trout in English.

Dam has an adverse effect in increasing mortality due to the prevention of migratory individuals from coming back. Just reducing the adult fish mortality rate by only 5% has a similar effect when the environmental carrying capacity K is halved (thin brown line in Fig. 15.5). The dam's adverse effect by closing the migratory paths that migrate during a lifetime is greater than the effect of habitat fragmentation. Especially, it has a great influence on diadromous fishes, ayu (*Plecoglossus altivelis*) and eels (e.g., *Anguilla japonica*) that migrate between the river and the sea. Some species, such as pink salmon (*Oncorhynchus gorbuscha*), all individuals go into the sea and 2 years later return to the same river to spawn eggs. In such species, they are only possible to deposit eggs in spawning sites downstream of the dam.

If dams prevent migration of diadromous fishes, the annual survival rate will decrease, as described above, and if the fish that went downstream does not return, there will be substantially the same effect as a significant decrease in the initial or juvenile survival rate. Furthermore, if the initial survival rate is halved, it will be almost extinct after 40 years, as shown in Fig. 15.5. If the initial survival rate has actually dropped significantly, it may be necessary to take measures such as establishing a fishway or even destroying the dam as soon as possible.

We have calculated the eigenvalues of the matrix using the geometric mean of the initial survival rate. As noted earlier, the maximum eigenvalue of the Leslie matrix in Table 15.3 based on the geometric mean of the initial survival rate does not match the long-term increase rate of population dynamics when the initial survival rate actually fluctuates annually. Therefore, we calculate the geometric mean of λ from the 90th root of the ratio N_{100}/N_{10}. As shown in Table 15.3, the average population growth rate per year did not match the maximum eigenvalue of the Leslie matrix

("Geometric mean of the maximum eigenvalues of *L*" in Table 15.3). Therefore, matrix evaluation using geometric mean or arithmetic mean is not always valid.

If the maximum eigenvalue λ is greater than 1 and if the carrying capacity K is large enough, the risk of extinction is very low. However, as shown in Fig. 15.5, even if λ is larger than 1 but if K is small, the extinction risk is not negligible. Perhaps the life table data in Table15.1 underestimates λ and the actual λ is much higher. For example, estimating the adult survival rate from the data obtained by examining the population when it is close to K leads to an underestimation of λ. The extinction risk of the char population, which should have persisted tens of thousands of years, cannot be so high. However, in terms of a larger spatial scale, extinction of local populations occurs frequently even in the natural state for centuries, and it is highly likely that they will be revived due to migration from other tributaries or rivers. Alternatively, there may always be a small amount of immigration in and out to support the survival of the population. If the dam existed for a 1000 years, it would have excluded the possibility of recolonization.

When local populations that have been separated to some extent in this way are maintained through recolonization, the entire population (set of local populations) is called meta-population. In the case of many freshwater fish, local populations exist as meta-populations for each river and are divided into small local populations by tributaries or by waterfalls or cascades. If there is a movement across a river, e.g., if some migratory individuals return to different river from their birthplace, we can consider a meta-population that includes fish in more than a single river. How to classify and understand the actual population structure is arbitrary, and the definition of the local population is also convenient. The same applies to the local division of ecosystems (Christensen et al. 1996). In addition, the concept of biological species itself is somewhat ambiguous.

There are some researches to estimate extinction risk by considering freshwater fish as a meta-population (Dunham and Rieman 1999). Theoretical studies have pointed out that there is a continuous increase in extinction risk as habitat fragmentation progresses, and there is no clear threshold (Jager et al. 2001). Anyway, the dam would be one of the major extinction risk factors that disrupt the habitat of freshwater fish and other aquatic diadromous organisms.

It is also known that the genetic diversity is lost by population fragmentation due to dams. It is not well known yet how it affects survival rate and reproductive rate, there must exist some effect.

If there is a fishway in some dams, fish may be able to go upstream. However, it is difficult to make a single fishway that is effective for salmonids, ayu, and diadromous invertebrates such as shrimp. In addition, most of the dams made in huge numbers in Japan do not have fishways. It is no exaggeration to say that Japanese rivers deprive the lifeline of natural fish and maintain it with released fish. Among the bivalves that live in the river, there are some species that attach to the salmon and trout gills and fins when they are larvae. If the host salmon and trout disappear, these bivalves will be extinct.

15.6 Are There Absolutely No Fish Upstream of the Dam?

Even if a dam is constructed, we sometimes find diadromous fish and shellfish that parasitize the diadromous fish upstream of the dam as shown in Photo 1 (Fig. 15.6).

There are at least two possibilities to explain this phenomenon. Shellfish have a long lifespan, and many live for several decades or longer. If young, small shells may not be found, and only large shells may remain in the upstream of the dam, there was probably no dams when they were born, and they will eventually become extinct. Some fish may be able to run up the dam. Migratory fish have the ability to ascend small waterfalls, but they cannot overcome the several meter cliffs of dams. The below picture was taken at a normal water level. If heavy rain causes the water level to rise, it may be possible to go up. In addition, there is a possibility of going up on a dam that is not so steep. However, we cannot accept a situation that there was a run-up case. As can be seen from the above computer experiments, even if the survival rate is lowered but positive, the risk of extinction increases significantly.

In a similar way to recognition of human life, we do not think the all-or-none in risks of ecosystems. It can tolerate a little load and may die casually. We do not forget the purpose of the Act on Promotion of Natural Regeneration, "considering that the ecosystem is maintained by a delicate balance." Nature is not always lost

Fig. 15.6 Dam in a certain place in Hokkaido (Photo by Kentaro Morita)

immediately even though a dam is created. However, their existence for decades or centuries will have a major impact. If we reflect on the past and destroy the dam, nature will have the power to regenerate even if it takes a long time. It is not too late.

15.7 How to Avoid Local Extinction Due to Dams

If the natural population growth rate λ is 1 or more and if the initial population is sufficiently large, extinction does not occur. However, it will be extinct if the carrying capacity declines substantially. As shown in Fig. 15.3, the risk of extinction within a century is not low enough.

Therefore, it is necessary to secure the natural rate of increase to 1 or more, preferably as much as the natural state, and maintain the environmental carrying capacity K to several hundreds or more. When K is smaller than the MVP, it is important that the "corridor" with the adjacent local population is not closed at least in the long term.

After experiencing the above calculations, readers may have realized that the estimation accuracy of life history parameters is low and that the extinction risk will change greatly even if the values are slightly changed. Moreover, the fluctuation range is large in order to predict the long-term population trend from the short-term survival rate estimation and the fluctuation of the population. In order to overcome these difficulties, conservation ecology emphasizes continuous monitoring, flexible changes in conservation measures based on monitored data (adaptability), and verification of the working hypothesis that are used in the management plan. This is called adaptive management.

Organisms have persisted through various catastrophic events since about 4.6 billion years ago. Nature itself has the ability to regenerate, and if we eliminate the obstacles that stop their lives, we can sustain nature's benefit to people in posterity. For that purpose, it is important to carry out ecological surveys such as population number and spatial distribution. We need to confirm that the population size does not decrease in the long term. Moreover, in the case of an isolated population, it is necessary to maintain the population size at least several hundreds, and when it is difficult, secure a corridor with the adjacent population. Moreover, if it turns out to be unsatisfactory in the long run, then it is necessary to modify the policy at that point in time to solve the problem. It is impractical to ban everything that is not proven safe. Such a manner is called "learn by doing," which is a famous slogan of conservation ecology. However, it is necessary to constantly monitor the population status during the project that may impact ecosystems. In addition, accountability to verify the hypothesis and adaptability to change the policy when it is found that the risk is high are indispensable when conducting a project.

15.8 Comparison with the Ecological Risk of Chemical Substances

The water quality standards of rivers are determined by risk assessment using a "fate model" that detects pathway and exposure level of chemical substances and "dose-response" relationship between the exposure level and human health risk through drinking water and food, as explained in Chap. 3. However, we now consider not only the human health risk but also the ecological risk (influence on the organisms living in the environment). The "indicator" organisms evaluated are mainly algae, plankton such as water flea (*Daphnia* spp.) and freshwater fish such as salmonids and medaka (*Oryzias* spp.). In order to investigate the chronic toxicity to fish, the effects on life history parameters such as hatching rate, initial survival rate, body growth rate, adult fish survival rate, egg production are examined. Sometimes we make life table data as shown in Table 15.1 and see the effects of exposure to chemical substances.

Although almost the same evaluation method as for human health risk is adopted in ecological risk assessment for chemical substances, not only the mortality rate but also the effect on reproduction is evaluated. However, direct assessment of the extinction risk of living organisms due to a chemical substance is rare. As shown in Table 15.3, the extinction risk depends not only on the mortality rate and reproductive rate but also on the number of individuals in the habitat area. Therefore, the permissible concentrations of chemicals should also vary with places and species. However, the standard for the concentration of chemical substances is generally decided nationwide. In order to effectively preserve nature, it is necessary to consider measures for each river while considering how much nature is left in each river.

In terms of health risk, the endpoint is human life or a person's quality of life, and the magnitude of a person's death hazard would be considered as equal irrespective of her/his age, health status, nationality, sex, income, intelligence quotient, and others. Perhaps such equality will be supported by society. In terms of ecological risk, it would be a social choice whether the extinction of chimpanzee (*Pan troglodytes*) and that of aye-aye (*Daubentonia madagascariensis*) are of equal value or not. There may also be controversy among taxonomists as to whether a taxon should be regarded as one species or two species, and the hazard of extinction may differ depending on the presence of related species. How to estimate the extinction hazards of local populations will be debated.

In the Environmental Impact Assessment Act, there are relatively clear quantitative standards for environmental chemicals, water quality, air pollution, noise, etc. Any business must satisfy to meet these standards in principle. However, the impact on threatened species and ecosystems should be as low as reasonably achievable (abbreviated by "ALARA"). If that is not possible, compensatory measures such as securing other habitats (off-site mitigation) will be taken. There is no clear numerical standard. Even in the Law for the Promotion of Nature Restoration, there is still no clear quantitative standard, and it is to be dealt with by adapting efforts through continuous monitoring. In other words, there is no guarantee that nature will be protected by a uniform standard, and continuous investigations will be conducted to monitor the ecosystem so that it will not be damaged and take necessary conservation measures.

Life expectancy is considered as an indicator of overall health risk. The life expectancy should be longer with decreasing mortality rate. New health risks, such as chemical hypersensitivity, are emerging. However, the average life expectancy is much lower now than it was a century ago. Death is biologically unavoidable, and the health risk due to each factor is probably set sufficiently low. However, we may not be optimistic about the indicators that cannot be evaluated by the endpoint of human life, such as human well-being and quality of life.

In contrast, the risk of extinction of living things is probably much higher now than it was one century ago. When I explained the MVP earlier, one of the criteria was that the extinction risk within the first century was 5% or less. Although we cannot force to stop any business whose impact is positive, we wish to stop the business if there is a real impact that one species out of every 20 species will be extinct after a century. “Learning by doing” is a basic method of adaptive management in conservation ecology. The level of ecological risk required for nature conservation is not an ideal, but merely an immediate and feasible goal. Seeking zero risk of biodiversity loss by 2020, as described in the Aichi Biodiversity Targets, is not feasible.

Regarding the health risk and the ecological risk, the relationship between the acceptable level and the risk level of the background level is different depending on whether it is handled by a uniform quantitative standard or qualitative goal. We can expect that regulation of pollutions that reduce the health risk will also reduce the ecological risk. However, regarding the concentration standard of chemical substances, we have come to evaluate the ecological risk with the same idea as the health risk. Nonylphenol, which has the potential to alter male fish in female, is an example regulated by an ecological risk rather than a health risk to humans. As a result, it was decided to impose strict concentration standards for chemical substances, while dams leaving a much greater impact on ecosystems as shown in this chapter.

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Chapter 16
Marine Comanagement Plan of Shiretoko World Heritage site

Hiroyuki Matsuda and Mitsutaku Makino

Abstract Shiretoko was inscribed as a Natural World Heritage site in 2005. It has an outstanding universal value as a connection between the terrestrial and marine ecosystems. However, coastal fisheries are operated throughout the area, and it was required that the protection of the area was strengthened during the nomination process. World Heritage areas are protected by the national laws of each country and are not under international control. Japanese coastal fisheries are based on comanagement of fisheries cooperative associations (FCA) aiming at sustainable fisheries. The fishers expanded the seasonal fishing-ban areas of walleye pollock (*Gadus chalcogrammus*), and Shiretoko became a World Heritage Site. In this way, Shiretoko became a case of a new world heritage, where the protection of nature was not guaranteed by the government but rather the initiative of the local stakeholders to protect it. Unlike other chapters, this chapter does not include explanation of mathematical techniques for ecological risk management. We discuss the importance of comanagement and decision-making by the local stakeholders in ecological risk management.

Keywords Bottom-up approach · Marine protected area · Fisheries cooperative association · sustainable fisheries · dams · Steller sea lion

16.1 Value of Nature in Shiretoko

World Heritage sites were selected on the basis of 10 criteria. Of these, the four listed in Table 16.1 are criteria for natural heritage. Shiretoko was evaluated as meeting both the (ix) "ecological processes" and (x) "biodiversity" of the criteria. At the stage

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Table 16.1 Criteria for Natural World Heritages (UNESCO 2019). (i)-(vi) are criteria for Cultural World Heritages

(vii) To contain superlative natural phenomena or areas of exceptional natural beauty and esthetic importance;
(iix) To be outstanding examples representing major stages of earth's history, including the record of life, significant ongoing geological processes in the development of landforms, or significant geomorphic or physiographic features;
(ix) To be outstanding examples representing significant ongoing ecological and biological processes in the evolution and development of terrestrial, freshwater, coastal and marine ecosystems and communities of plants and animals;
(x) To contain the most important and significant natural habitats for in situ conservation of biological diversity, including those containing threatened species of outstanding universal value from the point of view of science or conservation.

of nomination by the member state, criterion (vii) "natural beauty" was also considered to be relevant, but the reviewer did not accept that this standard was met. Yakushima, which was inscribed in 1993, meets the criteria (vii) and (ix), Shirakami inscribed in the same year meets the criteria (ix), and Ogasawara inscribed in 2011 meets the criteria (ix). To date, Japan has no World Heritage Site that meets the criterion (iix).

(ix) For the criterion of "ecological process," it is referred to "an outstanding example of the interaction of marine and terrestrial ecosystems as well as extraordinary ecosystem productivity, largely influenced by the formation of seasonal sea ice at the lowest latitude in the northern hemisphere."

(x) For the criterion of "biodiversity," it is referred to "particular importance for a number of marine and terrestrial species, some of them endangered and endemic, such as Blackiston's fish owl (*Bubo blakistoni*) and the *Viola kitamiana* plant. The site is globally important for threatened seabirds and migratory birds, a number of salmonid species, and for marine mammals including Steller's sea lion and some cetacean species."

In particular, the marine area was an indispensable element of the Shiretoko recommended world heritage site as a basis for satisfying the criterion (ix).

16.2 The Marine Area of the Nomination Site is a Fishing Ground

According to the World Heritage Convention, Member States are responsible for protecting the "outstanding universal value (OUV)" of their World Heritage Sites. Therefore, World Heritage sites are usually protected both legally and practically so as not to destroy nature. In Shiretoko, the terrestrial core area ("Area A") is protected by the Nature Parks Law and the Forest Ecosystem Protected Area scheme. The terrestrial buffer zone ("Area B") is designated as the second and third special areas

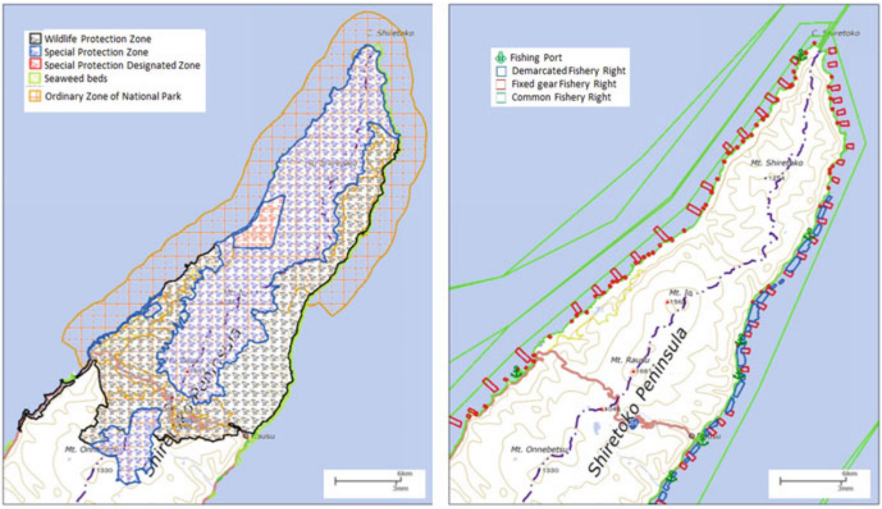

Fig. 16.1 Maps of Shiretoko World Heritage Site (left) and fisheries rights (right), drawn from Ocean status display system by Japan Coast Guard and Ministry of the Environment. There are several types of common fishery rights. The area inside the outermost green line is the common fishery rights area

of the national park, the conservation use area of the forest ecosystem protection area, and the green corridor. In addition, the marine area is designated as an ordinary area of Shiretoko National Park.

Set-net fishery and gill-net fishery in common fishery rights are conducted in the ordinary areas of national parks as shown in Fig. 16.1. The local Rausu fishers were worried that if the site was inscribed as a World Heritage site, their fishing activities were restricted. When applying for a World Heritage Site in the spring of 2004, the Ministry of the Environment and Hokkaido Prefectural Government made a commitment to the Fisheries Cooperative Association that “no additional fishery regulations will be imposed with the World Heritage.”

Immediately after applying for nomination, the government organized the Scientific Council for Shiretoko World Natural Heritage Site. Among the authors, Matsuda served as a member of the Shiretoko World Heritage Science Committee from 2004 to 2014 and Makino from 2014 to the present. A framework for formulating, implementing, and evaluating management plans was created based on the advice of scientists.

The review process of the world natural heritage is conducted by experts of the International Union for Conservation of Nature (IUCN) after the government nominates the candidate sites of natural heritage to UNESCO. Normally, the government officially invites IUCN auditors to the site. Judges can also get an impression of the possibility of registration by interviewing the media. As will be described later, in the case of Shiretoko, a great challenge happened during this process.

The World Heritage Committee usually meets every summer, the recommendations by IUCN are announced around May. There are four types of recommendations: inscribe, referral, deferral, and not to inscribe. In the case of a natural world heritage site, if IUCN recommends approval, it is to be approved by the World Heritage Committee. Even if the examination body recommends postponement of registration, the nomination country often works on member states to approve the inscription. Thus, due to the nature of the treaty, the final decision on the registration of a World Heritage Site is left to the voting of the Member States.

In the case of Shiretoko, IUCN has informally sought to further protection of the marine area around autumn 2004. The abovementioned promise by Hokkaido and the government to fishers and this IUCN's informal request seemed to be incompatible. Initially, this IUCN informal letter was not disclosed to the Scientific Council but was reported in the press.

The Scientific Council took the informal letter seriously and considered its role in discussing and advising how to respond to IUCN's requests. However, the government did not convene the Scientific Council and prepared a reply to IUCN without any advice from the Scientific Council.

Professor emeritus Kenkichi Ishigaki, who was the chair of the Scientific Council at that time, looked at the situation seriously. He compiled the opinion of the Scientific Council by email and sent it to the Ministry of the Environment. The Scientific Council expressed the view that two working groups, one for dams on rivers and one for marine, need to be created to address the IUCN requirements. However, the government disregarded the views of the Scientific Council and sent a reply to IUCN that no new fisheries regulations were needed.

Responding to this reply, IUCN has again urged, more explicitly, to sufficiently expand its marine areas and encourage the development of marine management plan.

There are also differences in speculation among UNESCO, the member state, local governments, and citizens. In addition to such a multilayered structure, the Ministry of the Environment, Forestry Agency, and Fisheries Agency may have different policies even within the government. Local governments also have a vertical relationship between prefectures and municipalities, as well as relationships between multiple municipalities that make up the registered site. There are various stakeholders in the area.

16.3 Scientific Council's "Extraordinary Advices" and Fishers' Decision

This time, the Scientific Council had a chance to advise it. The Scientific Council proposed that the fishers should take new protection measures in order to simultaneously meet the two constraints of "no additional regulation" and "strengthening protection levels" by IUCN. Fishers dislike top-down regulation because they have a tradition of managing marine resources by themselves.

Japanese coastal fisheries have the right to be exclusively used by local fishermen through the territorial user rights for fisheries (TURFs, Makino 2012) called common fishery right. This will prevent the tragedy of the commons (see Chap. 17) and lead to incentives for fishers to use their fishing grounds sustainably. In South American countries such as Chile, the artisan exclusive zone for small-scale fisheries are set inshore to encourage self-management by TURFs, while offshore waters are managed by topdown for industrial fisheries. As mentioned above, there are more than 1000 closed areas in Japanese fishing grounds (Yagi et al. 2010), but most of them are closed areas that have no legal collateral and are voluntarily set by fisheries cooperative associations. In addition to the fishing-ban areas, the FCA has established various input controls, such as limiting the fishing season and operating areas, and prohibiting the use of nets with a small mesh size to avoid catching small fish.

As pointed out not only in fisheries management, but also in Covid-19 (Chap. 6) and trace metal management (Chap. 4), Japan does not regulate with penalty, and the government often sets self-restraint for individuals and private companies. One reason for that would be probably a post-WWII legal system. Before the war, Japan had the Public Order Act, and the Emperor often issued an emergent edict to limit individual freedom. From that reflection, there is no emergency in the postwar Japanese legal system that limits the freedom of individuals with penalties. Rather, close communication and corporation between the government agencies and industrial organizations are highly developed. Therefore, the self-restraint or the autonomous management is often used as a tool for the industrial policy in Japan..

At Shiretoko, IUCN, an international organization, has sought to raise the level of protection because it was recommended by the international system of World Heritage. Moreover, it was probably due to the Steller sea lions. Sea lions are harmful to fishers because they break expensive nets and eat fish in the net.

Fishers could blame. Neither IUCN nor UNESCO has the power to interfere with domestic politics. However, they can reject the nomination. On the contrary, there should be a way to show that the fishers carry out while they protect the nature that is worthy of World Heritage. The Scientific Council did not force the fishers to increase their protection level. We expected that World Heritage inscription should be achieved when it also benefits the local community including fishers.

In the spring of 2005, the Rausu FCA voluntarily chose to expand the seasonal fishing-ban areas for walleye pollock (Fig. 16.2). It is a voluntary expansion, so it is not regulated by the government. This is not a legally protected area either. Therefore, once the walleye pollack resources are restored, it will not be necessary to continue fishing regulation.

Walleye pollack was one of Japan's major fisheries resources, reaching 3 million tons in the 1980s. However, due to the decrease in resources in the 1990s, the Rausu FCA decided to close eight fishing zones, including most of the spawning grounds, to the closed season at the end of March, which is the spawning season. In 2005, they decided to expand the number of closed areas by making six new fishing-ban areas. Now Japan has satisfied IUCN's request without breaking the promises to fishers. The government has not taken any concrete measures to raise the protection levels other than the compilation of the document "Marine area management plan." Since it expanded voluntarily, there is no compensation to fishers.

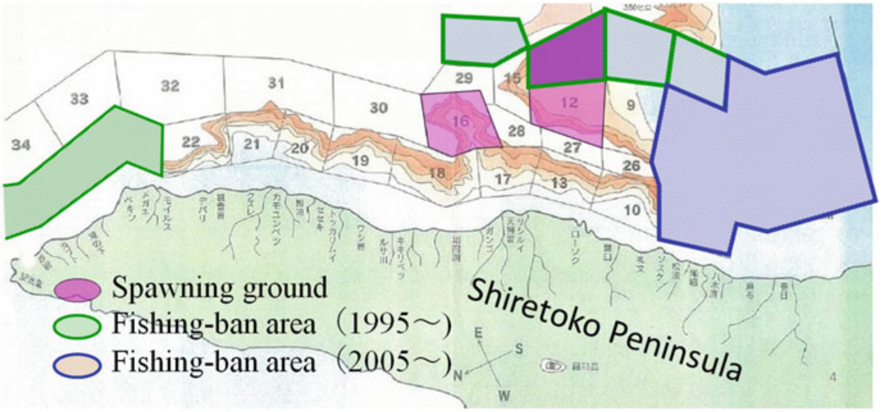

Fig. 16.2 Walleye pollack fishing grounds on the Rausu (eastern) side of the Shiretoko Peninsula (fishing areas numbered 1–34), spawning grounds (pink cells), and seasonally fishing-ban areas from 1995 and 2005 (green and blue cells). The marine area from the coast to the brown line is shallower than 200 m depth. Curves represent contour lines of depth

It is said that the majority of the members within the FCAs did not agree to the expansion of fishing-ban areas. However, due to the decision of the FCA's leader, the Rausu FCA decided to expand the seasonal fishing-ban areas for walleye pollock. At that time, the leader said that this was a voluntary initiative, not an expansion for World Heritage inscription.

IUCN recommended UNESCO to inscribe Shiretoko as a World Natural Heritage site in May 2005, and, congratulations, Shiretoko was registered as a World Heritage Site at the World Heritage Committee Meeting in 2005. In this way, fishers definitely played the greatest role to inscribe the Shiretoko World Heritage. This story was selected as one of the six " Impact Story" by the International Association of the Study of the Commons in 2010.

16.4 "Satoumi" in Japan

Three years after the Shiretoko World Heritage inscription, the 5th World Fisheries Congress was held in Yokohama in 2008. In 2010, the 10th Meeting of the Conference of the Parties to the Convention on Biological Diversity (CBD/CoP10) was held in Nagoya. Although it was suspected that criticism to Japanese fisheries would be received at these international conferences because Japan's fishery regulations were legally insufficient. In contrast, it was a good opportunity to introduce Japan's rich biodiversity and Japan's efforts of sustainable fisheries such as Shiretoko. Of the approximately 250,000 species of marine life in the world, 33,329 species have been found in Japan's Exclusive Economic Zone (EEZ), which is one of the world's most endowed marine biodiversity areas (Fujikura et al. 2010). Off Sanriku coast, the area where the giant earthquake attacked in 2011, is one of the "three major fishing grounds in the world" where cold and warm currents collide.

Under such favorable biogeographic conditions, the Japanese has ever been dependent on marine resources for a long time, and efforts to prevent overfishing have been made since ancient times. The Chronicles of Japan ("Nihon-shoki" or "Nihongi"), which was completed in 720, has a description of a fishing ban for small fish in Muko coast, Hyogo Prefecture around Nishinomiya City, in August 689 (Kagami 2012). The oldest marine protected area in the world is said to be the Royal National Park of Australia established in 1879, but the Japanese oldest MPA is much older than the oldest known MPA in the world.

There are a variety of objectives protected areas, from no-go and no-take zone of any natural resources to harmonizing various socioeconomic issues including fishery and tourism (Dudley 2008; Tsurita et al. 2018).

Satoumi in Japanese is defined by Yanagi (2019), but here we define it as a coastal ecosystem where people utilize natural resources sustainably. Even if the government does not legally control the catch amount, sustainable fisheries management could be achieved by the comanagement. However, it does not always work sustainably. In Japan, binding fishery management is not legally possible, as is Covid-19's lockdown. There are no other means than relying on voluntary management because legally binding management is impossible.

16.5 Initiatives of the Rausu Fishers

Walleye pollack was one of Japan's main fisheries resources during the 1970s. The Japanese pollack catch in the 1980s was ca. 500,000 tons, but it remarkably decreased in the 1990s and remained around ca. 200,000 tons in the 2000s (Ito 2019). Why has walleye pollack decreased?

Until 1985, Rausu FCA had 177 fishing vessels that caught walleye pollock. However, in the 1990s, the resource declined, and the catches drastically declined, and the risk of overfishing increased. However, the amount of most fisheries resources changes naturally over a period of several years to about 10 years due to climate change. However, even if the cause of the decrease is a natural phenomenon, it may be overfished if fishers continue to catch after the decrease. In 2004, the Rausu FCA reduced the number of fishing boats almost by half because the resources would be depleted.

Those who quit fishery definitely have to find another job. There will also be conflicts between those who quit and those who remain. The Rausu FCA paid compensation to those who quit (Makino et al. 2009). It is rare in the world for fishers to voluntarily compensate, not by the national or local government.

16.6 How to Prevent Overfishing

One of the reasons why overfishing does not stop is that fishing boats and nets are expensive. Fishery equipment costs are not in proportion to the number of operating days or the fishing effort. The cost of a boat or net will not be reduced even if fishers take a break from fishing. Also, the ship is damaged even if it is not operated. Rather, not using it would be short-lived, as same as a house or car.

Therefore, once fishers have fishing boats, they will want to continue fishing until the boats are abandoned, even if resources are reduced. However, many fisheries resources do not last for a long time as long as the life of fishing boats. If fishers continue to catch the resources even after the resources have decreased, the resources will decrease further. Even if the environment recovers, resources that have been exhausted cannot recover easily.

Approximately 1990, when overfishing of bluefin tuna became a problem, the Japanese government tried to reduce the number of tuna fishing vessels. However, the ship was sold to foreign countries and eventually continued to take tuna as a foreign flag of convenience. Instead, it was just out of the control of the Japanese government.

Agreement on compensation and reducing the number of ships is a great achievement. The captain of the sightseeing boat, which was taken care of by the Scientific Council when they visited Shiretoko, was also a fisher. We never felt they were willing to quit. Such transfers would not have progressed until resources were further reduced if the former fisher was not compensated.

In this way, the expansion of the seasonal fishing-ban area in response to IUCN's request was in part the fishers' voluntary efforts, such as the reduction of fishing vessels due to mutual aid compensation.

While inscription of Shiretoko as a World Heritage Site, the Scientific Council, explained this effort to the world in English (Makino et al. 2009; Matsuda et al. 2009, 2010; Makino 2012). The actual situation of comanagement of Japanese coastal fisheries was not well known abroad. Many consensuses are based on local discussions and are not recorded in government documents. Even overseas researchers could understand the official documents by the Japanese government to some extent, so it was known that there were few administrative measures, and Japan seemed to be in an illegal state of fishery management as much as possible. This would have been said about the differences between Japan and the European and American people in the Covid-19 emergency. But in reality, it was not an administrative measure, but a fishers' voluntary control.

The point is to prevent overfishing and maintain a sustainable society, not to make strict legislative or administrative regulatory instruments. There are many cases where citizens do not comply with the law. The history of the Shiretoko World Heritage nomination was a great opportunity to inform the world of the actual situation of fisheries comanagement in Japan (Makino 2012).

16.7 The Marine Management Plan in Shiretoko World Heritage Site

The Scientific Council and the Government have decided to develop a marine management plan in accordance with IUCN's recommendations. We have joined this work as a member of the Maritime Working Group of the Scientific Council. At that time, we proposed a policy of "drawing a blueprint of a house already built" (Matsuda et al. 2018). In other words, I thought that the mission of this management plan was not to impose new regulations but to clearly state what the fishery cooperative associations were already working on.

Another matter is to clarify the relationship between such fishing practices and the sustainability of marine ecosystems. For this purpose, we drew a "marine food web" that represents the predation relationship of marine ecosystems in Shiretoko (Fig. 16.3). We asked a student to draw a draft figure from the biota catalog of Shiretoko. Information about which species eats which prey is generally found in the international database FISHBASE to some extent. Experts on the Working Group revised it. The food web of the marine ecosystem includes terrestrial animals such as brown bears and sea eagles. Salmon return to the river and are eaten by brown bears. This represents the connection between the land and sea ecosystems that characterizes the Shiretoko World Heritage Site.

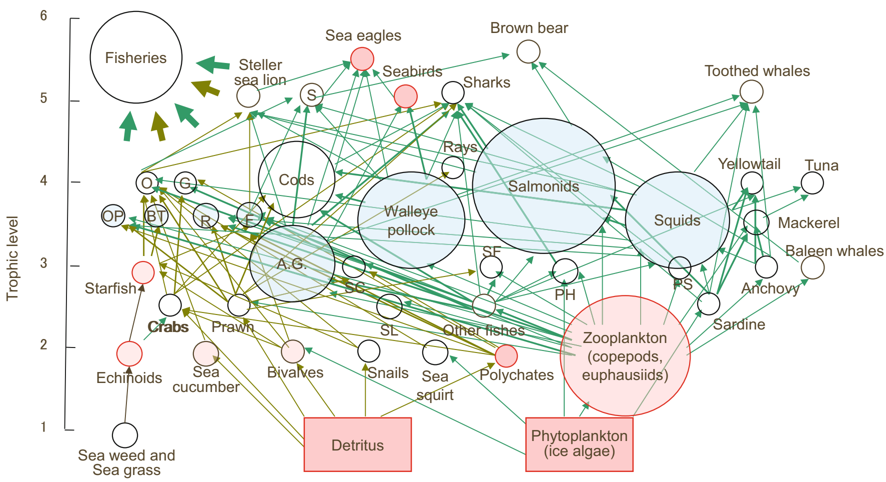

Fig. 16.3 Food web in the Shiretoko marine area (Source: Shiretoko World Natural Heritage Scientific Council). Circles and squares are both species and taxa, large circles have a large amount of resources, squares are not used by humans. Arrows indicate the relationship from prey to its predator. AG: arabesque greenling; BT: bighand thornyhead; F: flatfishes; G: greenlings; O: octopus; OP: ocean perch; PH: Pacific herring; PS: Pacific saury; R: rockfish; S: seals; SC: saffron cod; SF: sandfish; SL: sandeel (Matsuda et al. 2009)

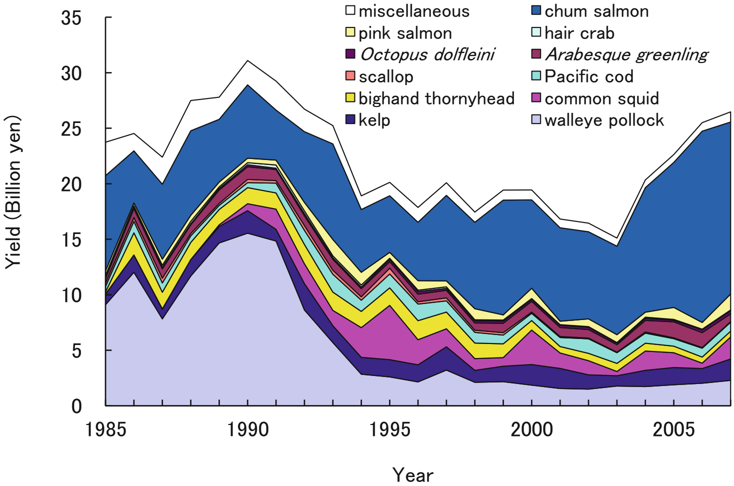

Fig. 16.4 Annual change in yield in the Shiretoko waters (Rausu and Shari, Drawing from the Current Status of Hokkaido Fisheries)

We can see that most living things are used by people. Some resources are not caught by fishers, such as brown bears. Those used by humans have almost complete annual catch statistics.

The catch statistics include both the catch amount (by metric tons, C) and yield (by Japanese yen, Y) of the landed fish. These are materials that show how many fishing resources have been acquired and how they have changed. Marine trophic index (MTI) was used as an indicator of overfishing, which is defined as the sum of trophic levels of all fisheries catch. The MTI decreased in some regions, which implies overfishing (fishing down, Pauly et al. 1998). Japanese fishery did not show a long-term decline in MTI (Matsuda et al. 2009).

Of these, Fig. 16.4 shows the annual change in the yield. As mentioned earlier, fisheries resources change naturally. Even in Shiretoko, walleye pollack and salmon were the main fisheries resources until the early 1990s, but salmons have increased since then, and squid (*Todarodes pacificus*), kelp (mainly *Saccharina japonica*), Arabesque greenling (*Pleurogrammus azonus*), etc., have become the second-largest catch by year. They use many fish species, but we can see that economically the top few species make up most of the total catch.

The yield is the product of the catch amount and the average fish price, and the catch amount is not the resource amount in the sea. However, with the catch amount and the yield, we can get some information about the ecosystem. Especially when we use many fish species like Japan and collect the statistics, we can get more information about the ecosystem. Information obtained from fisheries is valuable for knowing the state of the marine ecosystem and for conservation (Matsuda et al. 2009).

Just writing down what fishers have been working on in the marine management plan does not mean that nothing will change. Shiretoko's efforts will be seen by people all over the world if they are highlighted by the world as a good practice case study of World Heritage. There are no areas where there is nothing wrong but we often explain only good things when we submit it to UNESCO.

Shiretoko was registered as a World Heritage Site in 2005, but the locals were not happy as it was. At the time of approval of nomination, the World Heritage Committee urged to solve the dam problem and the overcrowding problem in the future and to formulate a marine management plan that clarifies measures for strengthening marine conservation and the possibility of further expanding the marine area. These responses were not usual.

Mr. Waki of Rausu Mayor said when he heard this news, "there was a real concern that Rausu's fishers might be aware that the fishing regulations would increase repeatedly" (The Yomiuri Shimbun, June 2, 2006).

The body that nominates the World Heritage Site is not the locals. World heritage is defined by the international convention, and the government of member state determines and nominates sites, and the government is responsible for responding to recommendations by the World Heritage Committee after inscription. Therefore, even if the conditions of the IUCN Recommendation are met by the voluntary efforts of fishers, it is the responsibility of the government to continue to fulfill the outstanding universal values of Shiretoko World Heritage. If fishers do not agree with the tightening of self-regulation, the government will have a negative legacy.

The World Heritage Site is required to report to UNESCO every 5 years after the inscription and receives various recommendations from the World Heritage Committee each time. There is a recommendation at the time of inscription. The government will explain in its regular report how it responded to the recommendation. Some of these recommendations often impose a new burden on the site. There is a list of "heritages in crisis" that may lose their value as world heritage sites. The criteria for "heritage in crisis" are also defined. Galapagos has once been put on the list of heritage in crisis due to problems such as over-tourism. It was removed from the list in 2010. Heritage in crisis requires a report every year. For world heritage sites, the World Heritage Committee is held every year and many international environmental groups also hang up there.

When an expert from the IUCN salmon specialist group visited Shiretoko in 2004, he pointed out that dams in Shiretoko prevent the salmon from going up. He wrote in a newspaper that he "hoped dams in Shiretoko as a model case for policy change on dams in Japan" (Yomiuri Shimbun June 15, 2005, in Japanese). Inscription of a world heritage has both benefits burdens in the locals.

Based on the 2005 recommendations, in February 2008, two representatives from IUCN and UNESCO visited Shiretoko. At that time, we explained about the mass capture of deer at the innermost part of the World Heritage Site (Cape Shiretoko). They almost agreed. The outbreak of deer is a big concern all over Japan. Deer eat plants and give big damages to natural vegetation and farms. This is a reason that we need to exterminate deer for the purpose of protecting natural vegetation in Shiretoko. At the dinner party, we ate deer meat in front of them. However, exterminating and eating sea lions did not convince them (Chap. 13).

16.8 An Excellent Model for the Management of Natural World Heritage Sites

In addition to the government, the Scientific Council and staff of the fisheries cooperative associations gave explanations to the Reactive Monitoring Mission Team in February 2008 Fig. 16.5. The Ministry of the Environment officials placed on the side of the mission team. They look happy with our explanation. This is probably the decision of the Ministry of the Environment. It was possible to impress that the local people played a major role in the management of the world heritage site.

According to the report of the mission team, "The mission noted the good progress made by Japan in protecting the Shiretoko World Heritage site and, in particular, addressing recommendations from the 2005 World Heritage Committee and the 2005 IUCN Evaluation Report. The mission was particularly impressed by the strong commitment of stakeholders at all levels to ensuring the Outstanding Universal Values of the property are maintained and passed intact to future generations. This is well reflected in the Shiretoko World Treasure Declaration signed by the Governor of Hokkaido and the Mayors of the two local towns, Shari and Rausu, in October 2005..." "The mission team also applauds the bottom up approach to management through the involvement of local communities and local stakeholders, and also the way in which scientific knowledge has been effectively applied to the management of the property through the overall scientific Committee and the specific Working Groups that have been set up. These provide an excellent model for the management of natural World Heritage Sites elsewhere" (Rao and Sheppard 2008).

This is one of the biggest compliments. From this time, we can say that Shiretoko has become one of the best practices from the recognition that it almost failed in the review process. After that, the Ministry of the Environment established scientific councils in Ogasawara, which is listed for the fourth nomination, as well as the

Fig. 16.5 Mr. Lao from UNESCO and Mr. Sheppard from IUCN (center left and right in the front row), the Reactive Monitoring Mission Team, listened to our presentation (February 5, 2005, photo by H.M.)

existing sites of Yakushima and Shirakami. Matsuda was invited as a member of the Scientific Council for the Yakushima World Heritage site.

The abovementioned report by the mission team emphasizes the bottom-up efforts of local stakeholders, rather than the top-down nature protection by the government. This came to be called the “Shiretoko Approach” (Makino et al. 2009). The Scientific Council for Japanese world heritage was established for the first time in Shiretoko, but it has spread to other heritage sites. Originally, the world heritage was not only a top-down system but rather a two-sided dynamic flow of decision-making between the government and the NGO level, such as the Rice Paddy Resolution of the Ramsar Convention for Biodiversity Conservation Practice, December 14, 2016.

The reason why it became possible to inscribe a World Heritage Site through the strengthening of autonomous management by the local fishers was probably that Shari Town had a long history of enthusiastic efforts to protect nature, such as the trust movement called “Shiretoko 100 Square Meters Movement.” It can be said that, as an extension of the efforts they have made so far, there was a ground to answer the IUCN recommendation at the time of recommending the World Heritage Site.

The trial continued even after the inscription. In 2012, a resolution to the World Heritage Committee was proposed to remove the dam on the Rusha River (Image of underwater concrete removal of dam at the mouth of the Rusha River (IUCN 2020)). The Scientific Council also had the opinion that the dam would need to be removed, but because there are facilities downstream, it is unacceptable to remove the dam and cause disasters. The removal of the dam should be possible if the downstream facilities are removed. In fact, the salmon trout hatchery in the downstream was removed in 2012. Local and government are trying to resolve it over time. International organizations rush into the local consensus building process and disturb it. Their purpose is to insist on justice, not to make it happen. Fortunately, with the understanding of other countries, it was amended into a resolution appraising Japan’s effort in 2012.

In 2018, the Scientific Council agreed to cut down the dam on the Rusha River so that salmon can go up and cars can cross (Fig. 16.6). The minimum structure has remained. We do not say that this is the removal of dams or bridges, but the important thing is to secure opportunities for salmon run-up and minimize impacts on the ecosystem.

There is a fishing base for fishers near the estuary of the Rusha River. Wild brown bears wander around it, but the fishermen “scold” the brown bears. They had no gun or bear spray. It is a miracle to coexist with armless fishers and wild bears, which was broadcasted by a NHK program on December 24, 2018. Similar phenomenon is seen on Kodiak Island in Alaska. If the World Heritage Committee respects this miraculous coexistence, they may agree with the removal of bridges sooner. The coexistence between people and wild bears has never been evaluated in the checklist of natural world heritages. There is no guarantee that this relationship will be sustainable.

Fig. 16.6 Image of underwater concrete removal of dam at the mouth of the Rusha River (Rand 2020)

16.9 Does Shiretoko Change World Heritages?

The natural world heritage site regularly reports the situation to UNESCO every 5 years. Looking at the checklist, there is no positive rating on using natural assets. Sustainable use will not be negative, nor getting a higher rating than not using it.

Biosphere reserves promoted by UNESCO's Man and the Biosphere (MAB) Programme and UNESCO global geoparks are part of the natural reserve system alongside the world heritage sites. Biosphere reserves and global geoparks are required to have valuable ecological elements and geological heritage, respectively. At the same time, both of them will be evaluated for participatory approaches by local stakeholders. It is different in character from natural world heritages.

However, neither was doing it differently from the beginning. When the MAB Program was launched in 1971, the emphasis was on protecting the biosphere and its use in educational and research activities. From around 1995, the strategy changed to protect the precious core of nature and establish a transition area around it to achieve sustainable use. The doughnut-shaped structure is the basic form of biosphere reserves, with a buffer zone in the middle so that no contradiction occurs between the protection of nature in core areas and sustainable use of natural assets in transition areas.

Some sites, such as Yakushima in Japan, Jeju Island in South Korea, and the Galapagos Islands in Ecuador, are designated as both natural world heritage and biosphere reserves. In that case, the core area of the biosphere reserve is usually designated as a world heritage site. In other words, only the part that should be protected is inscribed as a world heritage site, and both parts that should be protected and utilized are included in a biosphere reserve.

Just as the characteristics of world heritage sites and biosphere reserves had changed in the past, it may change in the future. Shiretoko, which includes fishing grounds in the world heritage site, may not be typical of the current world heritages. However, it may become a model for future world heritage, as the mission team highly praised. Both biosphere reserves and global geoparks have chosen the path that encourages the sustainable use of natural assets over time.

UNESCO and IUCN must also be looking for success stories. In addition to being strict, we will find significance in developing UNESCO activities and developing conservation movements around the world under world heritage systems.

It is more important to nurture people who are responsible to protect nature than to protect nature itself. If the government is legally binding to protect nature or if the international convention forces the locals to protect nature, there may be a conflict with the local people. It is not a good idea for governments and environmental groups to develop a nature protection movement. It is important for the inhabitants to understand that nature conservation secures people's lives of the next generation and brings people's happiness. If nature conservation does not necessarily lead to human well-being, it is not necessary to protect nature. The purpose of the Convention on Biological Diversity is to maintain sustainable human well-being and to maintain the nature's contribution to people or ecosystem services.

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Chapter 17
A Guideline for Ecological Risk Management Procedure

Hiroyuki Matsuda

Abstract This chapter proposes practical guidelines for ecological risk management. Recommended procedures include screening for potential ecological risks, scoping relevant stakeholders and management areas, defining "undesirable events" as endpoint, identifying risk factors, assessing probabilistic risks, and building a risk management plan. It includes management strategic evaluation methods, planning, consensus building, and review process of the management. At that time, there are a part that can be scientifically advanced and a part that should be socially agreed. Therefore, it is important to reach a social consensus at the stage of agreeing the management purpose and the stage of agreeing the management plan.

Keywords Adaptive management · Screening · Scoping · Monitoring · Consensus building · Climate change · Land use change

17.1 Basic Procedure of Ecological Risk Management

Environmental policies include global issues such as climate change mitigation measures and local issues such as nature reserve management. The latter may contribute to global issues such as greenhouse gas reduction, but aims to make a direct contribution to the local community. The latter is expected to have a direct effect on the local environment by the management measures implemented by the agreement of the local community.

Ecological risk management procedure consists of environmental risk assessment, management, and process of consensus building with risk communications. In other words, the procedure is shown in Fig. 17.1.

Involving local citizens and stakeholders into the management of ecosystems is often recommended (Millennium Ecosystem Assessment 2005). Local environmental issues usually include the global context, which is characterized by the slogan

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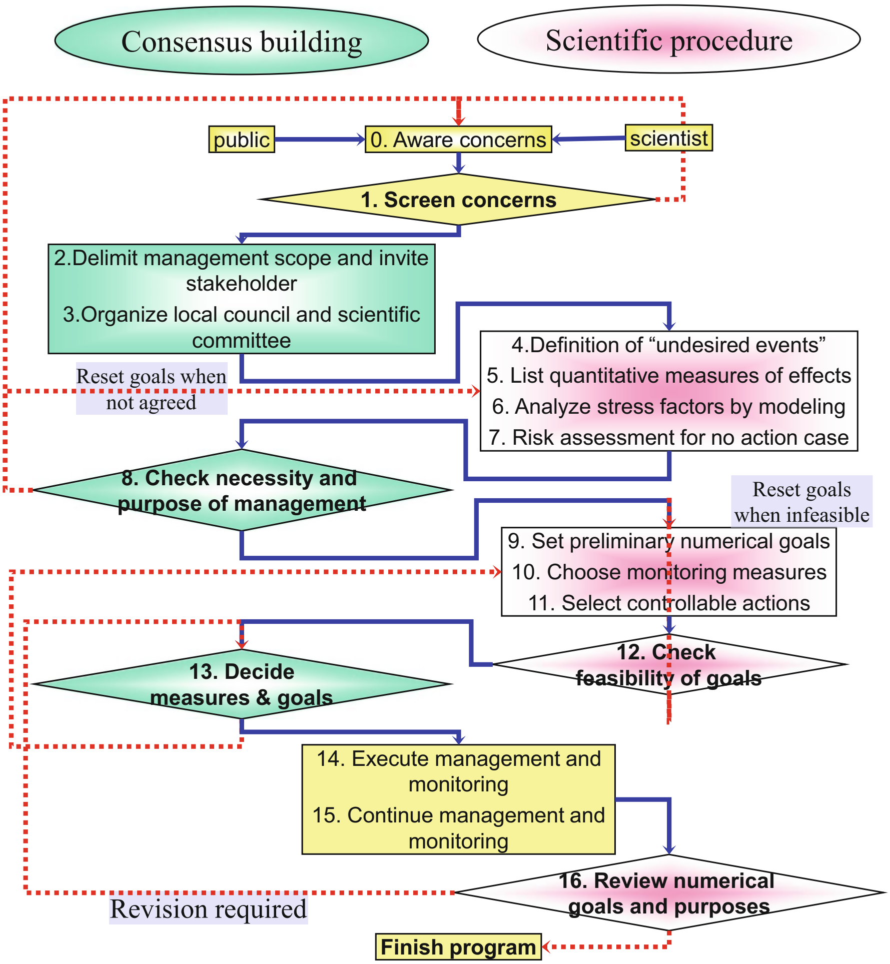

Fig. 17.1 Basic procedure of ecological risk management (Rossberg et al. 2005)

"Think globally, act locally!." But technical, organizational, and political barriers can stand in the way of local involvement, in particular, if the management has to deal with a situation of scientific and public uncertainties (Rossberg et al. 2005).

In this chapter, we try to clarify theoretical considerations and practical experiences with local ecological risk management projects that may be unclear in initializing local ecological risk management procedures. This does not mean to encourage more active involvement of the scientific community (Rossberg et al. 2005). We will describe step-by-step how a project is led.

Matsuda et al. (2005) also addressed the guideline for nature restoration projects, which includes the idea of risk management in one of the 26 principles. Christensen et al. (1996) gave principal ideas of ecosystem management, whose many points are common with ecological risk management. We can distinguish at least four tasks:

(1) getting the relevant stakeholders involved, (2) attaining a basic scientific understanding of the problem, (3) reaching agreement on a risk management plan, and (4) the implementation of the plan depending on monitoring results, which requires adjusting targets and methods. These tasks are connected by a system of feedback loops and two decision points.

As shown in Fig. 17.1. we recommend to perform (7) “Risk assessment for no action case” in order to clarify what happens under business as usual before (8) “Agreement of management purpose.” In order to carry out a risk assessment, (4) definition of “undesired events” or “endpoints” in the context of risk science, (5) “list of quantitative measures of effects” and (6) “analysis of stress factors by building quantitative models” are indispensable. However, such procedures are not established that this is done before the agreement of management purpose. In that sense, the “risk evaluation in the case of management” is also performed in (12) the “check feasibility of achieving the target.” In addition, scientific proposals for feasible numerical goals and management implementation plans are made before social consensus is reached.

The flow shown in Fig. 17.1 is the basic idea, and risk management is flexibly planned and implemented according to the actual situation. The first point is to establish a series of risk assessment procedures prior to the agreement of the management purpose. The second point is to obtain social consensus by stakeholders in at least two stages: agreement on purpose and agreement on management plan with numerical goals. The third point is to include adaptive risk management by the implementation of management, monitoring results, and reviewing the plan.

17.1.1 Initiation by Social Demands or Scientific Concerns

The target of risk management becomes a social problem due to social demand or scientist warning. When sika deer increased too much and the damage to agriculture and forestry increased, population control of deer was due to the demand of society. When scientists find that tributyltin used for ship bottom paint sterilized all species of snails, regulation of tributyltin was agreed by scientists’ initiative (Smith 1981).

It is not always the case that these issues are directly subject to risk management. As a result of preliminary risk assessment, it may be determined whether particular measures need to be taken or not. This process is called screening. This may be triggered by the fact that countermeasures are taken after a major incident actually occurs and the incidents that presage it are widely reported. In any case, the media have a great influence. There are various ways in which an issue is addressed.

17.1.2 List of Stakeholders and Management Scope

When risk management is enforced, some must list the stakeholders involved in the problem. It also defines the extent to which the problem will be addressed. For example, in the case of the bird collision problem of a wind power plant, are we concerned only for birds or also bats? Do we assess avian collisions in combination with the landscape and noise problems of the residents or not? Should the discussion be limited to the narrow area planned by the business operator or choose appropriate places for building wind power stations? The development of the discussion may change depending on the scope. This corresponds to scoping in environmental impact assessment.

Since environmental issues are interrelated, if you take a deeper look, the stakeholders will endlessly expand. The broader it is, usually the more difficult it is to solve a real problem. Stakeholders can be narrowed down if the problem is limited, but there may be a backlash from excluded people. It is desirable to capture the range of stakeholders involved in consensus building as broadly as possible within a realistic range. Stakeholders may be added as necessary thereafter. However, we should note that those who join later may not comply with the earlier agreement. It is also important to disclose information at an early stage and secure opportunities for stakeholders to participate. In short, it is important to steadily build a relationship of trust by avoiding the situation where consensus building is redone (Levin 1999).

It may not be needed to go through all of the steps in this section if there are no serious conflicts between stakeholders. These procedures are effective means for fully preparing for uncertainties, as well as necessary for the rational resolution of conflicts among stakeholders. Although misunderstood especially in Japan's environmental impact assessment, the procedure is not a means to prevent business operators from making anything as they like. It is also a means for conflict avoidance that facilitates consensus building for business operators as well, by following these procedures. This can also be associated with the basic matters relating to the guidelines to the Environmental Impact Assessment Act (Ministry of the Environment, Japan 1997) and international framework conventions on environmental issues.

17.1.3 Establishment of a Council and a Scientific Committee

A formal decision-making body for stakeholders (hereinafter referred to as the "council") and an advisory organization by scientists (hereinafter referred to as the "scientific committee") are established to formulate a management plan, It is necessary to analyze the monitoring results after the implementation of management, review the management plan, and give advice to the management body regarding how to respond to the opinions by the stakeholders and the public. At that time, the purpose and contents of the management project should be clearly stated. The local and external stakeholders, relevant administrative agencies, intellectuals, environmental groups should be widely invited. It is important that “a wide and fair opportunity for participation should be secured” (Ministry of the Environment, Japan 2003).

The council is responsible for formulating, deciding, monitoring, and amending the management plan. In principle, the decision is unanimous, but in international negotiations, various rules may exist. It ensures the freedom of the members to speak, the requirements for the establishment of important projects (for example, approval of more than 3/4), the right to reserve, and the method of withdrawal. Publishing the minutes of councils and scientific committees, leaving the materials as official records, and then publishing them will be an effective means of avoiding conflicts. In addition, it is possible to enhance the convenience of those who wish to browse by utilizing the internet. It is important to ensure transparency through such information disclosure and opportunities of involving the public comments.

These are the means to facilitate the implementation of the management plan upon agreement.

17.1.4 Scientific Definition of Undesired Events

Undesired events are called endpoints in risk science. This is not something that can be determined solely by social demands. For example, when deer grows too much and eats agricultural crops in the field and peels the bark of forest trees, damage to agriculture and forestry becomes a social issue. Deer, on the other hand, are indispensable members of the ecosystem from biodiversity viewpoints. It is necessary, including scientists and the general public, to clarify the background of the social problems to consider comprehensive solutions. To that end, it is necessary to scientifically organize the undesired phenomena.

When probabilistically expressing a risk, it is necessary to clearly define the event so that it can be objectively judged whether the event occurs or not.

17.1.5 List of Quantitative Measures of Effect

We clarify indicators that can objectively evaluate the state of events or probabilistic risks that should be avoided in order to achieve the objectives and address the identified problems. If we like to avoid “loss of biodiversity,” for example, the number of Steller’s sea eagle individuals is set as an assessment indicator. We may have more than one assessment indicators. Various indicators can be considered in advance for a variety of management goals.

17.1.6 Analysis of Stress Factors by Modeling

Undesired events are exposed to various stress factors. Scientists analyze these factors and consider the countermeasures. For example, factors such as habitat degradation, overfishing, environmental pollution, invasion of exotic species, and climate change may be the factors that cause the extinction of populations. Furthermore, overfishing may involve not only fisheries that directly target the organism, but also bycatch of other organisms, and dumping that is discarded as useless at sea. In this case scientists predict how the evaluation indicators will change due to these factors. The prediction is usually associated with uncertainty. The future of quantitative evaluation indicators can be predicted by constructing a mathematical model for prediction considering risk.

Stress factors include those that are controllable, difficult to control immediately, and uncontrollable. For the case of fisheries management, we can control catch quota but cannot control the sea surface temperature in the next year. It is necessary to distinguish between them.

17.1.7 Risk Assessment Under Business as Usual

We evaluate the risk when no special measures are taken or “business as usual.” If this risk is not acceptable, we need to adopt some measures to reduce the risk. What is the business as usual depends on the case. For climate change, RCP8.5 is not the business as usual but the worst scenario (Hausfather and Peters 2020). We can say that the RCP8.5 scenario in which the temperature will rise by 4.5 degrees is no longer BAU because of progress in climate change measures. Nature is resilient, so if the environmental impact is small, there would be no need for aggressive restoration measures. Passive restoration, which takes advantage of the resilience of nature, is also recommended when adopting conservation measures (Matsuda et al. 2005).

17.1.8 Agreement of Necessity of Management and Purpose

When the actual situation of the risks becomes clear to some extent, the necessity of risk management is recognized and the purpose is socially agreed. It is desirable to agree on the objectives when the location of the problem becomes clear and before concrete measures are proposed. If concrete measures are defined, the interests of each stakeholder will be overt, and the objectives may be distorted by other interests than the overall objectives.

Even if another agreement is needed at the decision of concrete measures, it is not preferable that the management purpose is influenced by the immediate profit. However, what is agreed at this stage is abstract content such as the management philosophy and purpose, and as a general rule, it is limited to the content that all stakeholders can agree.

The purpose may be an abstract idea, such as "conservation of biodiversity." It makes sure that as many stakeholders as possible can agree. In the future, if there is a possibility that this purpose will not be clearly met in the management business, it will be the basis for reviewing the management plan. In the above example, rather than simply setting the management objective of preventing damage to agriculture and forestry by deer, it would be possible to solve the problem from a comprehensive perspective.

In order to engage in constructive discussion, it is necessary to assume the existence of stakeholders with diverse values, we avoid binary confrontation and find a solution that allows more stakeholders to convince. Since the ecosystem is an open system, it is necessary to have a management plan that considers a wider area, but at the same time, it is important to focus on the problems that can be solved locally.

17.2 Proposal of the Draft Risk Management Plan

17.2.1 Setting of Draft Numerical Goals

After stakeholders agreed to the necessity of the management, scientists set draft numerical goals that can be objectively judged to be successful in the future, and establish monitoring items for verifying them. Numerical goals are set by some deadline. For example, the stock recovery plan of southern bluefin tuna was agreed with the numerical goal of "recovering the 1980 stock abundance level by 2020." Due to uncertainty in the estimated stock abundance, we have set an unwavering target for future improvements in estimation methods, such as the "1980 level," rather than an absolute number agreement with three million tons. Moreover, it is effective to set goals in the near future that can be reviewed every few years, rather than goals in the too distant future.

We need to examine the feasibility, including risk assessment and cost-effectiveness, of achieving the numerical goals under the feasible management plan. The following series of procedures is needed to judge whether the numerical goals set are appropriate or not. Therefore, it is necessary to set a draft numerical goal and confirm its feasibility.

The number of years to reach the numerical goal varies, and can be considered from several weeks to 100 years from the present. However, it is necessary to periodically evaluate the achievement level and review the plan described later. For example, assess achievement every year or every few years. If it is necessary to adjust policies based on annual evaluations, it is desirable to incorporate them in the original plan so that feedback can be made in advance. However, since it is possible that the initial plan may not be met, evaluations should be conducted every few years, including a review of the plan, and efforts should be made to improve the feasibility of longer term goals.

17.2.2 Choose Monitoring Measures

It is desirable to decide in advance which method will be used to evaluate state changes during monitoring. Suppose that there is a numerical goal that the population size of a particular endangered species exceeds a certain level, if an estimation method of the population size is not agreed, we cannot decide whether the target is achieved or not. This may be necessary especially if the stakeholders are in serious conflict.

Risk assessment is not always based solely on scientifically verified assumptions. The risk of unforeseen events can vary greatly depending on the assumptions of the assessment method. In such a case, the scientific committee has to discuss which assumption is used, and if there are disagreements, both the committee should describe both assumptions and evaluate separately by multiple assumptions. Revising the assumptions by reviewing the results of monitoring is called adaptive learning, which is an important part of adaptive management. It is also desirable to determine in advance what kind of result should be obtained and how to review it.

17.2.3 Selection of Controllable Items and Actions

After setting numerical goals, we need to plan the management measures in order to achieve the targets. We estimate the feasibility from scientific and realistic viewpoints. Some risk influencing factors are controllable by the management entity and some are not. In addition, there is a method that can be implemented only by the agreement of the public by publicizing the risk assessment. For example, the possibility of killing wildlife, especially wild monkeys, depends on social consensus. We should not choose measures that cannot achieve the targets. We should select the management method required to achieve the numerical goals. The risk assessment method may not be limited to established methods and may be freely determined according to the management purpose, assessment indicators, and numerical goal. However, the scientific validity of the method is thoroughly examined through an objective and scientific external evaluation. The scientific results are announced, and the public participation by the opinion submission procedure is done after the agreement of the public scientific committee.

17.2.4 Feasibility Study of Achieving the Targets

There is always a risk that the numerical goals are not achieved because the management plan involves uncertainty. This corresponds to the risk assessment when a certain policy is implemented. If the achievement is difficult, it is necessary to show the rationale for the difficulty and review the numerical goals. In other words, it is necessary to evaluate the feasibility of achieving the goal before implementing the risk management plan. The evaluation also includes economic and social constraints. If it is difficult to realize the situation, revise the plan by changing the provisional setting of concrete goals.

A mathematical model called an operating model is sometimes used to examine the feasibility and problems from future monitoring to decision-making procedures, especially. In fisheries management, such a comprehensive scheme for risk management is called "management strategy evaluation" or MSE (Punt et al. 2014). Either of these makes it possible to assess risk under various control measures.

Risk management is carried out by defining "monitoring," "risk assessment method," "numerical goals," and "management measures."

The magnitude of risk in a situation of large uncertainty becomes considerably large when the once-determined management policy is implemented regardless of the subsequent monitoring results. However, if the management policy is flexibly changed in the future according to the monitoring situation, it is possible to take a conservation measure before it falls into a serious situation. Such management is called "feedback control," which significantly reduces the risk. In addition, it is necessary to repeatedly verify and review the unverified assumptions used for risk assessment by monitoring after implementation of the management plan. In other words, adaptive management consists of two parts, feedback control and adaptive learning, which reduces the risk.

17.2.5 Decision of Risk Management Plan and Their Numerical Goals

The management plan including management measures and evaluation methods are not decided by scientists alone but are socially agreed. Therefore, the scientific committee should reflect the will of the council, such as preparing alternative targets and measures, and the council and the scientific committee should refine the management plan for each other.

After agreement of the management plan, risk assessment and risk management methods in the Scientific Committee and the Council, and before implementing the management, inform the stakeholders and the general public. It is necessary to carry out the procedure for submitting opinions and to form consensus based on the opinions of stakeholders and the general public. This procedure may be carried out twice, at the management purpose and the management plan. In this way, it is often smoother to engage in stakeholder involvement and to reach consensus. The more difficult it is to reach an agreement, the better it is to build step-by-step consensus. If we proceed to the next step without an agreement, there is a high risk that discussions will be overwhelmingly traced back to the stage before the agreement.

If it is not possible to agree, it is necessary to reexamine the management plan including numerical goals. This will be repeated until a scientifically feasible and socially agreed management plan is developed.

17.3 Implementation and Monitoring of the Management Plan

While implementing the management plan and continuing monitoring, it is necessary to verify the unverified assumptions used for risk assessment. In other words, the adaptive management is a hypothesis verification experiment. Depending on the result, the assumptions should be changed promptly, the risk assessment method should be reviewed, and the management plan should be revised if necessary. This is the accountability of the manager.

17.3.1 Review Process of Achievement of Numerical Goals and Purposes

In this way, it is desirable to perform management while demonstrating the assumptions used. The extent to which the proof can be achieved varies, and when it is possible to prove that excludes other hypotheses (testable). Even if the alternative hypothesis cannot be eliminated, if the result of management is unexpected, it means that the assumption is incorrect (falsifiable).

Monitoring is indispensable for adaptive management. The items to be surveyed are determined by the evaluation method in advance, but in some cases, it is necessary to agree on the method of conducting the survey and the person in charge, and mutual monitoring of the survey itself may be necessary. In some cases, the business side and the environmental group distrust the investigators who are recommended by the other party. Even in such a case, the conflict may be resolved if both investigators survey together. This is called “joint fact-finding” (Matsuura and Schenk 2017).

During the implementation of the plan, we should verify whether the numerical goals are feasible and whether there are any unexpected problems. Even if the numerical objectives are met, it cannot be said to be successful if there are critical problems in light of the management objectives.

17.3.2 Review of Management Plan

As a result of conducting management and evaluation, when an unexpected situation of the management plan occurred, we analyze the cause and consider whether it could not be estimated in advance, and review the management plan if necessary. To that end, continuing monitoring and management planning is crucial. When making a major revision, in principle, the same procedure for consensus building is newly redone. When conflicts are expected, it is necessary to decide in advance when the consensus should be redone, as described in Japan's Environmental Impact Assessment Act.

Management plans do not always last forever. It may be possible to revise it every several years, and once it achieves or fails to meet its intended purpose, it may stop and the efforts of stakeholders may be reassigned to new problems. Conversely, if the target is achieved, it may be possible to set a higher target and review the management plan. In addition, it may be integrated with the daily economic activities of stakeholders.

17.4 The Tragedy of Mitigation Measures in Climate Change Issues

There are two types of countermeasures to climate change, mitigation, and adaptation. If the main cause of climate change due to human activities is the increase in greenhouse gas concentrations in the atmosphere, reducing greenhouse gas emissions ensures that climate change is mitigated. Emissions are global in nature and are determined by the combined emissions of all countries. It does not matter who issued it If country A reduced emissions and country B increased, climate change impacts would be equally on both countries.

On the other hand, adaptation measures that reduce the adverse effects of climate change often benefit the nation or the local entity who adopt them. Although each country or each entity works on both mitigation and adaptation measures. Here, think about which and how much they should serve their effort. What is often considered is the optimal solution that maximizes the global net benefit. However, in reality, each country pursues its own benefits and can free to choose its own policies. In that case, it is considered to be in the situation of noncooperative game theory.

The cost of each country to invest in adaptation measures and mitigation measures is regarded as a strategy, and the benefits of each country are expressed as a function of each country's adaptation costs and mitigation costs. Suppose the profit V_i of country i is expressed by the following formula.

$$V_i = B_i \exp[-D_i(\Sigma M_j, A_i)] - M_i - A_i$$

where M_i and A_i are amounts of investment in mitigation and adaptation measures of country i; B_i is the economic size of country i, D_i is the deterrence effects of climate change impacts which is a function of the mitigation measures of the world and adaptation measures of that country. We assume that D_i is an increasing function of the total mitigation investment in the world ΣM_j and the adaptation investment in the country i. However, net income is $B_i - M_i - A_i$, the gross benefit minus investment in mitigation and adaptation measures.

The Nash solution or noncooperative equilibrium satisfies the following equations:

$$\partial V_i / \partial M_i = -(\partial D_i / \partial M_i)(V_i + M_i + A_i) - 1 = 0$$

and

$$\partial V_i / \partial A_i = -(\partial D_i / \partial A_i)(V_i + M_i + A_i) - 1 = 0$$

On the other hand, the Pareto solution or cooperative equilibrium set to maximize ΣV_j satisfies:

$$\partial \Sigma V_j / \partial M_i = -\Sigma(\partial D_j / \partial M_i)(V_j + M_j + A_j) - 1 = 0$$

and

$$\partial V_i / \partial A_i = 0.$$

For the sake of simplicity, we suppose there are only two countries and the following functional forms (Brechet et al. 2013).

$$D_i(M, A_i) = \mu(M) g(A_i) = \left[\eta_L + (\eta_U - \eta_L) e^{-bM}\right] / (1 + g_0 A_i)$$

If $(b, g_0, \eta_U, \eta_L, B_1, B_2) = (2,2,0.9,0.1, 90,70)$, we numerically obtain the Pareto solution:

$$(M_1, A_1, M_2, A_2, V_1, V_2) = (0.86, 0.87, 0.86, 0.60, 84.2, 64.7)$$

In other words, mitigation efforts in both countries are equal.

The Nash solution is

$$(M_1, A_1, M_2, A_2, V_1, V_2) = (2.42, 0, 0, 1.58, 78.5, 66.7).$$

That is, the country with smaller B_i does not invest in mitigation measures. In this case, the larger country invests a big effort in mitigation measures. The total profits of both countries must be less than the Pareto solution.

This is the same situation as the tragedy of the commons discussed in the chapter on fishery management. As the number of countries increases, the total mitigation investments of all countries decreases and more emphasis will be placed on adaptation. For each country, it is best for other countries to invest in mitigation measures for the world, and for themselves to invest adaptation measures.

There are several ideas to avoid the tragedy of the commons. One idea is to divide the common properties into private properties. For example, in the far seas fishery, vast high seas have existed in the Pacific, Atlantic, and Indian Oceans. Since the UN Convention on the Law of the Sea came into effect in 1994, coastal resources became available exclusively to coastal countries up to about 200 nautical miles. However, greenhouse gas concentration is homogeneous in the world and cannot be divided. In the fishing industry, there are also fish species that migrate across exclusive economic zones. Second, it may be possible to adopt the optimization policy to maximize the total net benefit as shown above, in which any nations are not allowed to seek their own benefit. The Kyoto Protocol adopted in 2007 was similar to this situation. We have set global emission reduction targets to reduce the greenhouse gas concentration and assigned them to each member state of UNFCCC. However, COP15 of UNFCCC in Copenhagen in 2009, aimed at an agreement following the Kyoto Protocol, revealed that no new legally binding agreement is possible. The weakness of the Pareto solution is that even if the optimum value of global emissions is obtained, it is not decided how to allocate it to each country. In other words, even if the total ΣM_j of mitigation investments is determined, the M_i of each country cannot be determined. Therefore, Pareto solutions are often called "Pareto sets."

The third way to avoid the tragedy of the commons is to build mutual aid partnerships. When describing the tragedy of the commons about fisheries, we referred it to the iterated prisoner's dilemma. By playing the game repeatedly with the same neighbor, the mutually beneficial cooperation strategy of "cooperating unless the neighbor betrays" can become a Nash solution (or strictly describing "collectively stable," Axelrod 1984). In the chapter of Shiretoko, by recognizing the territorial user rights of the coastal fishers, it becomes easier for self-regulation such as setting fishing ban areas to function without being punished by law mutually beneficial cooperation will be more easily established. Finally, it is possible to internalize external diseconomies of environmental impacts, such as by trading greenhouse gas emissions. This is a variant of the second idea of the global order to force cooperative action in each country. Cap-and-trade system sets the emission limit in the world but allows market transactions for its allocation between countries or between organizations. Although disputes can arise when deciding on the initial allocation, we expect that it will be easier to agree than the second idea because greenhouse gas emission limit is not fixed and can be traded.

The UNFCCC invented another way, after failing to agree on a legally binding mitigation measure. That is the "nationally determined contribution" (NDC) described in the next section. This can be interpreted as an incentive for realizing mutually cooperative partnerships.

17.5 Ecosystem Impact Assessment of Climate Change and Its Countermeasures

Climate change is one of the most important global environmental problems. Both emphases will be placed on mitigation measures to reduce greenhouse gas (GHG) emissions and adaptation measures to reduce the adverse effects of climate change. Climate change is concerned not only for human health but also for industrial production and biodiversity. However, it has been pointed out that if biofuel farmland is expanded significantly for mitigation measures, such mitigation measures themselves will result in habitat loss of wildlife and damage biodiversity.

In the IPBES (Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services) related to the Convention on Biological Diversity, the main factors that impair biodiversity are (1) land use change, (2) climate change, (3) overexploitation, (4) exotic species, and (5) pollution (IPBES n.d.). In addition to these, the Japanese Ministry of the Environment (2010) directed the 6th factor of biodiversity loss as the reduction of human activities (Ministry of the Environment, Japan 2016).

The IPCC uses four representative concentration pathway (RCP) scenarios for GHG concentrations (IPCC 2007), called RCP8.5, RCP6.0, RCP4.5, and RCP2.6. For example, RCP8.5 means that the "radiative forcing" at the end of this century is 8.5 W/m^2, which means that the global average temperature will increase by about 4.5 degrees (median forecast) compared to before the Industrial Revolution. RCP2.6 is a scenario in which temperature rise is said to be suppressed to about 2 degrees, and RCP4.5 and RCP6.0 are scenarios in between.

IPCC also established five shared socioeconomic pathways (SSP), taking into consideration measures such as land use change to achieve the mitigation goal, SSP1 is Sustainable, and SSP2 is Moderate that means mixture of the other four scenarios. SSP3 is called Regional division, SSP4 is called Disparity, and SSP5 is called Fossil fuel dependence. SSP1 falls within RCP6.0 even without any additional mitigation measures. SSP5 will correspond to RCP8.5 or 4.5 degrees increase if it does not take mitigation measures such as carbon dioxide capture and storage (CCS). Land use change such as biofuel cultivation also differs for each combination of RCP and SSP.

Of the five factors that impact biodiversity considered in IPBES, only climate change and land use change are considered in the IPCC scenario analyses, but we can predict future biodiversity loss using quantitative models under each scenario.

From the information of the current distribution of each animal and plant species, the environmental conditions such as weather and vegetation where each species inhabits are calculated. This is called a species distribution model (SDM). If future climate and vegetation change under climate change scenarios used in the IPCC, the potential habitat areas of each species will change, and extinction risk will also be calculated. At that time, in consideration of the migration speed of each taxa such as plants and birds, birds may expand to new habitats, although plants may not shift their habitat. Thus, future predictions of biodiversity loss are made from climate change scenarios and SDM.

Hof et al. (2018) showed that RCP2.6 suffered more biodiversity loss than RCP6.0. They predict that land use changes for biofuel cultivation will outweigh the adverse effects of climate change. However, they did not compare with RCP8.5 and did not analyze plants or reptiles.

In contrast, Ohashi et al. (2019) showed that RCP2.6 has less biodiversity loss than RCP8.5. Both use the same RCP scenarios and SDMs but differ in the method in more detail. These analyses suggest that RCP8.5 has a big impact on biodiversity but RCP6.0, which increases the atmosphere temperature by 3.0 °C. Ohashi et al. (2019) analyzed mammals, reptiles, and vascular plants. According to Ohashi et al. (2019), RCP8.5 loss in plants is particularly significant in North America, Asia, Oceania, and Europe.

By continent, there is no significant difference between RCP2.6 and RCP8.5 scenarios in Europe and Oceania. The adverse effects of climate change without mitigation measures were largely offset by land use changes due to mitigation measures. Each SSP scenario can be seen that the adverse effects of the social fragmentation scenario of SSP3 in Africa are larger than those of other SSP scenarios in both climate change and land use change.

Not only in biodiversity but also in many papers, the negative impact of RCP8.5, which is estimated to increase by about 4.5 degrees without taking mitigation measures, is higher than in other climate change scenarios. RCP4.5 and RCP6.0 scenarios, which keep the rise around 3 degrees, are less affected than RCP8.5.

In addition, the impact depends on the climatic zone, such as subtropical zone and subarctic zone. Although grain production decreases as a whole, climate change at high latitudes is expected to increase yields. Iizumi et al. (2018) showed that the yield model estimated the impact of climate change up to now on the world average yields of corn, wheat, and soybean over the past 30 years (1981–2010). Red and green show a decrease and an increase in yield due to climate change, respectively; white is an area where no grain is grown. Increased production of rice in Japan is expected.

Under the UN Framework Convention on Climate Change, all countries must work together to tackle climate change. In reality, there are regional differences in their adverse effects.

As is often said about global environmental issues, developed countries and developing countries have different responsibilities to the past GHG emissions and different contributions to climate change. The 1992 Rio Declaration used the term "common but differentiated responsibility." The aim of the international convention is sustainable development, and it does not hinder the economic development of developing countries.

However, isolated responses of each country would be undesirable. At the time of the Kyoto Protocol of the 2007 UN Framework Convention on Climate Change, GHG reduction targets were assigned to each developed country with a great effort by comprehensive efforts of the Secretariat of the Convention and stakeholders. However, the Paris Agreement could not agree on such top-down goals (Rogelj et al. 2016). Instead, the Paris Agreement seeks to ensure an effective response to the issue of climate change by ensuring that each country promises a “nationally determined contribution (NDC)” and assessing its achievement. It may be the intention of eliciting a positive attitude in each country by officially indicating the contribution. Following the Kyoto Protocol, only the superpower can leave the Paris Agreement without worrying about the unpopularity of other countries.

As mentioned above, the north-south difference exists both in GHG emissions and the severity of climate change impacts. We recognize such unevenness in impacts and responsibilities and we also recognize that climate change is a problem common to all humankind. How to fulfill different responsibilities is required. Even if the NDCs of each country are implemented, the target is not actually achieved, and it is predicted that it will increase by about 3 °C by 2100. However, the NDC based on the Paris Agreement is a 5-year plan from 2020, and each country will make a contribution in the next 5 years, so we can expect that GHG reduction will be strengthened in the future. Furthermore, there is uncertainty in the forecast of temperature rises and the impacts of climate change on society.

Therefore, even if the current NDC does not reach the 2 °C or 1.5 °C target, it is sufficiently meaningful to avoid the RCP8.5 scenario, which is expected to increase by about 4.5 °C and there is a big reason to set even more reduction targets.

Moreover, global environmental issues go beyond mere environmental issues and are incorporated into the 17 Sustainable Development Goals (SDGs) for more comprehensive sustainable development. The adverse effects of climate change are not limited to the environment but include health and industry, and comprehensive adaptation measures will be taken to avoid a variety of adverse effects. In that sense, climate change adaptation measures are compatible with the SDGs.

Thus, climate change is not the only global environmental issue, and environmental issues should not be prioritized over peace, hunger, and health. However, we can say that the IPCC presents a framework that covers the entire SDGs, including adaptation measures. Preventing climate change is a means to achieve the SDGs effectively, not an ultimate goal. That is why the issue of climate change is recognized as one of the most important for global environmental issues.

17.6 Social Demands/Raising Scientific Problems

The basic flow of risk management procedures aims to make it easier to smoothly proceed with consensus building by dividing into three stages of abstract purpose, verifiable numerical goal, and management plan to execute it. In the case of serious conflicts, stakeholders often make conclusions before making a dispute. However, it is difficult to clarify the “opponent’s fault” in the decision process of making abstract purpose. It is possible to kick the seat, but it is less likely that the action will get the support of the public opinion compared to kicking the seat after clarifying the opponent’s nonsense. In this way, the method of consensus building that agrees on the purpose without defining specific regulatory measures and standards, then establishes specific methods of implementation, monitoring, and evaluation. As in the international conventions on climate change and biodiversity, the purpose is written in the article of the treaty, and the implementation plan is set in the protocol.

Such consensus building procedure is difficult when some of the stakeholders only seek for the cancelation of the plan. However, even in that case, if some party kicks off without any rational excuse, the rest of the parties will be able to reach an agreement. The legitimacy is ultimately determined by public opinion. If someone breaks down for an irrational excuse, the broken side will be criticized by public opinion. Therefore, it will not be easily broken at the stage of determining the initial idea. If, after agreeing on the objectives, they refuse to agree on numerical goals or implementation plans, there will be a moral obligation for the refusing party to present a feasible alternative. All stakeholders will be required to make efforts to reach consensus. Each counterproposal will be examined by the scientific committee for feasibility and concerns, and the council will discuss the results based on the monitored data. Even if the scientific committee reports that it is difficult to realize, the responsibility of the person who forcibly carries out the management will be criticized.

Opportunities for citizens to express their opinions are being secured, such as the enforcement of the Act on Access to Information Held by Administrative Organs and the establishment of the public comment system decided by the Japanese Cabinet in 1998. The latter is to secure the opportunity to reflect citizens’ diverse opinions on the administrative policy, to make the process of policy formulation more transparent, and to fulfill the accountability of the government. It is also considered that the publication of documents became technically easy with the spread of new information media such as the Internet, which also contributed to the spread of public opinions. Given that there is volatility in the purpose of nature conservation and evaluation criteria, such consensus building procedures are more important. The 12 principles of the ecosystem approach (Convention on Biological Diversity Secretariat 2000) respect scientific knowledge, conventional knowledge and traditional knowledge. A transdisciplinary approach that does not rely solely on scientific knowledge is recommended.

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Chapter 18
Multispecies Fisheries Management

Hiroyuki Matsuda, Peter A. Abrams, and Toshio Katsukawa

Abstract Maximum Sustainable Yield (MSY) theory, based on a single species dynamics model, has been considered to rule out both no-take rules and overfishing. It is thought to recommend moderate catch rates that do not lead to exhaustion of the fish population. However, ecosystems are complex systems, and they usually have uncertain observations and nonstationary dynamics even in the absence of fisheries or interspecific interactions. There is a need to build a new management theory that takes these aspects of an "ecosystem approach" into consideration.

One example, where single species approaches are clearly inappropriate is the case of sardine, anchovy, and chub mackerel. It is known that these small pelagic fish repeatedly alternate outbreaks every several decades, and this phenomenon is called "fish species replacement". The mechanism of species change has not been clarified, and it is difficult to predict its dynamics.

Adaptive management is attracting attention as a strategy for sustainable resource use based on feedback control and adaptive learning with uncertain information (Walters 1986). Adaptive management is a measure to understand resource dynamics through continuous monitoring and to flexibly respond to fluctuations of resource abundance. Rather than predicting resource fluctuations, we aim for human beings to respond appropriately ex post facto. Adaptive management can cope with resources that are difficult to predict and have large fluctuations.

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In previous work, we also considered how to design fishery regulations in systems having various prey/predator types. This work assumed that the amount of effort towards catching each fish species could be adjusted independently. The fishing effort for each species together with their abundances and relative values defined the total fisheries yield, denoted by Y, which is called the "multispecies MSY" or the maximum sustainable revenue. We selected 1000 virtual food webs with random variables of community structure parameters and calculated the multispecies MSY for a system with coexistence equilibrium points in the absence of catch. We also asked for MSY (called conservation MSY) under the constraint that all species persist.

Keywords Maximum sustainable yield · Maximum sustainable ecosystem service · Ecosystem approach · Target switching · Adaptive management · Feedback control

18.1 Introduction

Wildlife management and fisheries management must consider the uncertainty of stock assessment, the nonstationarity of the dynamics of natural systems, and the complexity of natural ecosystems due to interactions between species. Nonstationarity can occur in single species systems, but it is known to be more common in multispecies system. We begin by addressing how to design management under such nonstationarity. Next, we consider sustainable yield obtained from multiple fish species. The idea that maximum sustainable yield does not result in species extinction in the use of single species resources does not hold true in the larger systems in which at least two species interact. In general, the MSY of one exploited species depends on the abundance of the other species. The MSY for the whole system can sometimes be increased by the eradication of a less valuable species. In this study, we investigated the multispecies MSY that maximizes the total yield obtained from the multispecies system using a mathematical model. Population fluctuations are possible in multispecies systems due to environmental fluctuations, but also due to the inherent instability of the ecosystem itself. The classical theory of Maximum Sustainable Production (MSY) does not apply to any of these cases.

Katsukawa and Matsuda (2003) verified the effect of a target switching strategy on fisheries that protect currently rare resources as a management policy for nonstationary fish species. The idea is to respond to unpredictable fluctuations of fisheries stocks ex post facto by switching the target of fishing efforts depending on the stock status at that time.

Next, we consider the sustainable yield obtained from multiple fish species. The idea, based on single species models, that maximum sustainable yield does not result in species extinction does not hold true in an ecosystem where at least two species interact. The MSY of one exploited resource often depends on the abundance of another resource species. If that other species is not valuable, the system's MSY can be increased by eradication of that species. In this study, we investigated the multispecies MSY that maximizes the total yield obtained from the multispecies system using a mathematical model.

In addition to the target switching strategy and multispecies management, we developed the resource recovery plan of the Pacific mackerel stock revealed from the start of this project in 2003. Since then, the Fisheries Agency, Japan, has implemented restrictions on the harvest of chub mackerel. In addition, the criterion for the fishing ban was included in the decision rule of allowable biological catch (ABC).

18.2 A Target Switching Strategy for Fisheries

In fisheries, ecosystem management differs from population management in that the catch rate for a given fish species can be changed by the conditions experienced by other species, including their rate of harvest. As a simple example, we will introduce the idea of a target switching strategy.

As mentioned above, the abundance of small pelagic fish varies greatly. In addition, when using multiple fish species with interspecific interactions, basing harvest on distinct population dynamic models for each fish species is not always effective.

As the simplest method for managing multiple fish species, we examined the effect of a switching harvest strategy that protects species when they are rare (Matsuda and Katsukawa 2002; Katsukawa and Matsuda 2003). This is a catch policy that changes the fish species used depending on the stock status at that time. Therefore, the appropriate catch for a given fish species depends not only on that fish species but also on the abundance of other fish species.

We consider, for example, the following fishery community model:

$$\frac{dN_i}{dt} = \left(r_i(t) - \sum_{j=1} a_{ij} N_j - f_j(t) \right) N_i \tag{18.1}$$

Here, N_i, r_i, and f_i represent the abundance of the species i, its natural increase rate at time t, and the per capita catch rate, respectively. The parameter a_{ij} represents the strength of the interspecific competition measuring the per capita effect of species j on the rate of increase of species i. We also assume that r_i changes with time.

In general, we consider a constant harvest ratio (CHR) policy to be one that keeps the catch rate of each fish species regardless of the stock abundance. A constant harvest amount (CHA) policy is one that keeps the catch amount constant. A wide range of temporally variable harvest strategies are possible; these normally involve a catch rate that depends on the current stock abundance. One of these is the constant escapement strategy (CES), which aims for a constant stock abundance after the harvest is obtained using optimal control theory under a fluctuating environment. The catch quota, $C(t)$, is determined by:

$$C(t) = fN(t) \text{ under CHR,}$$
$$C(t) = \mathrm{Max}[0, N(t) - N_c] \text{ under CES,} \tag{18.2}$$

where N_c is the target stock abundance after the fishing season.

Here, we consider a CHR policy and a switching catch for a two-species system in which the fishing rate on species i is expressed as follows in proportion to the stock of species i:

$$f_i(t) = \frac{f_0 N_i(t)}{\sum_j N_j(t)} \tag{18.3}$$

where f_0 is the sum of the fishing efforts for the two fish species.

We considered the three cases, (i) two species that fluctuate independently, (ii) a competitive system (Katsukawa and Matsuda 2003), and (iii) a three-species competitive system with a cyclic advantage relationship leading to fish species replacement (Matsuda and Katsukawa 2002). In all three cases, the target switching strategy is obtained based on all fish species that are present. We have theoretically shown that this switching strategy has the effect of increasing the total catch and of increasing the minimum stock level during the regime of stock decline.

Figure 18.1 shows an example of a mathematical model in the case of both periodic variation (affecting the growth rate parameter of the two species undergoing fish species replacement in the opposite manner) and short-term random variation in that parameter:

$$r_i = r_0\left[1 + r_a \sin{(-1)^i}\frac{2\pi t}{T} + r_e \xi_e(t)\right] \tag{18.4}$$

where r_a and T respectively represent the magnitude and period of the environmental change; r_e and $\xi_e(t)$ represent the short-term random environmental change and its amplitude by a uniform random variable between -1 and 1.

Figure 18.1 shows the difference between the target switching strategy and constant harvest rate (CHR) policy. The abundances of these two species naturally fluctuate every decade, and species 1 and 2 are out of phase by 5 years. The catch rate ($f_0 = 15$) in this figure is set to be relatively high. In the target switching strategy, the stock abundance does not decrease much in the low season, and the total catch of the two species is larger than that in the CHR policy. However, the variation in species composition of catches is large. However, if the total catch rate that achieves the maximum sustainable catch of a single fish species is kept ($f_0 = 10$), the difference between the target switching strategy and the CHR policy is not very large. As shown in Fig. 18.1, when the catch rate becomes excessive, the difference between

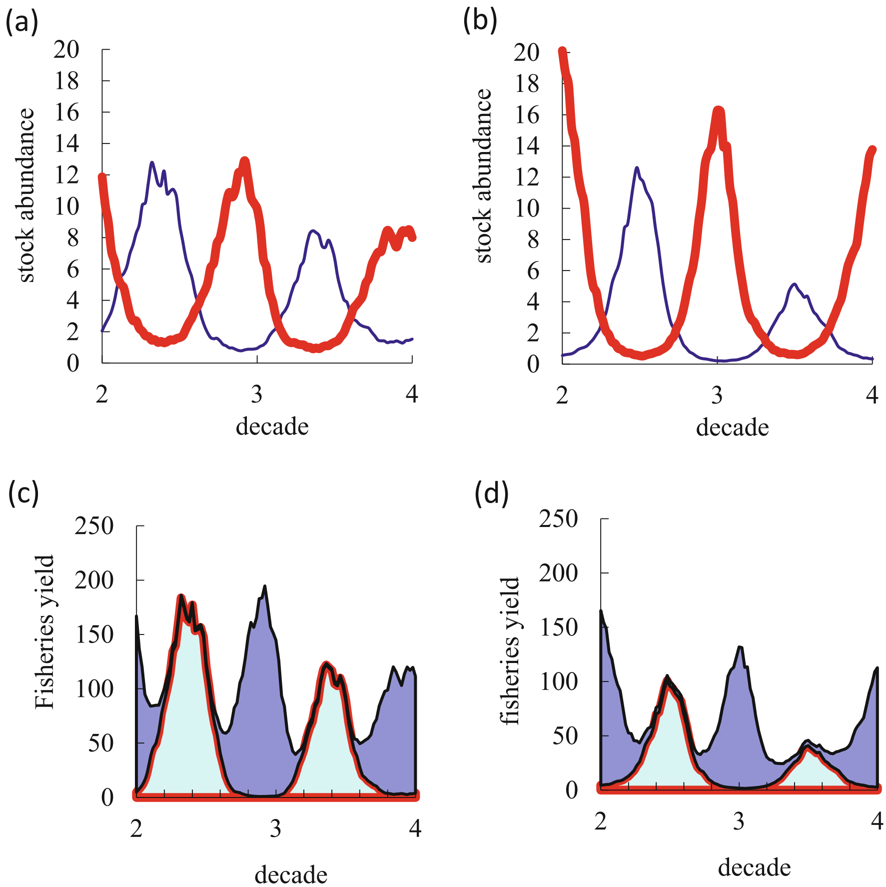


Fig. 18.1 Fluctuation in the stock abundance and fisheries yield with switching (**a**, **c**) and no-switching ($f_1 = f_2 = 0.5$) (**b**, **d**). These are based on a two-species mathematical model. $a_{11} = a_{22} = 0.2$, $a_{12} = a_{21} = 0.1$, $r_0 = 10$, $r_a = 0.5$, $r_e = 0.5$, $T = 1$, $f_0 = 15$. The same initial values and $r_i(t)$ are used in the left and right hand figures (Katsukawa and Matsuda 2003)

the two becomes significant, and the target switching strategy has the effect of increasing the minimum stock abundance.

A potential disadvantage of target switching strategy is that the catch of each species varies with a larger magnitude than the stock. However, the target switching strategy is effective in preventing stock depletion due to the combined effects of overfishing and natural fluctuation. In addition, switching has the effect of increasing and stabilizing the total yield.

Since we published these articles in 2002, no country has adopted a target switching strategy in actual fishing policy, even though an ecosystem approach has repeatedly been recommended during that time period.

18.3 Maximum Sustainable Yield from the Multispecies Community

As noted above, real ecosystems are characterized by uncertainties, nonequilibrium dynamics, and species interactions. The classical theory of maximum sustainable yield (MSY) ignores community interactions; uncertainty in stock assessments, ecosystem processes, and process errors in population dynamics. It also ignores intrinsically and externally caused fluctuations in population densities (Hilborn 2002; Matsuda and Katsukawa 2002). All of these factors are indispensable in understanding ecosystem processes. Their absence from applied management explains at least in part the relative lack of success of fisheries management (Hannesson 1996; Matsuda and Abrams 2008).

On the other hand, ecosystem approach and ecosystem-based fisheries management have long been popular slogans. Here is one example. According to Article 14 of the Kyoto Declaration and Action Plan on Sustainable Contribution of Fisheries for Food Security in 1995 hosted by the Government of Japan and sponsored by FAO, "When and where appropriate, consider harvesting multiple trophic levels in a manner consistent with sustainable development of these resources." However, in the two-species system of prey–predator, the solution that maximizes total catch does not result in the simultaneous use of both species (Clark 1985). In order to mathematically examine the conditions for using fish species with various trophic levels, Matsuda and Abrams (2006) considered the following food web consisting of up to six species.

$$\frac{dN_i}{dt} = \left(r_i - \sum_{j=1}^{s} a_{ij} N_j - q_i E_i \right) N_i \tag{18.5}$$

where N_i is the stock abundance of species i, r_i, and a_{ij} are the intrinsic rate of population increase and the per capita interspecific effect of j on i; a_{ij} is positive if j consumes i, and negative if positive if i consumes j. The parameters q_i and E_i are the fishing efficiency and fishing effort on species i. The equilibrium state, denoted by N_i*, is the solution of the simultaneous equations in the parentheses given the restriction that N_i cannot be negative for any i.

Total yield Y at the equilibrium is obtained by

$$Y = \sum_{i=1}^{s} p_i q_i E_i N_i^* \tag{18.6}$$

where p_i is the price of species i. We call the total yield Y when the fishing effort E_i is set to maximize Y as the "multispecies MSY" or simply "MSY." Matsuda and Abrams (2006) obtained the "multispecies MSY" by considering 1000 hypothetical systems consisting of six species with randomly selected parameter values under the assumption that a coexistence equilibrium (positive abundance of all six species)

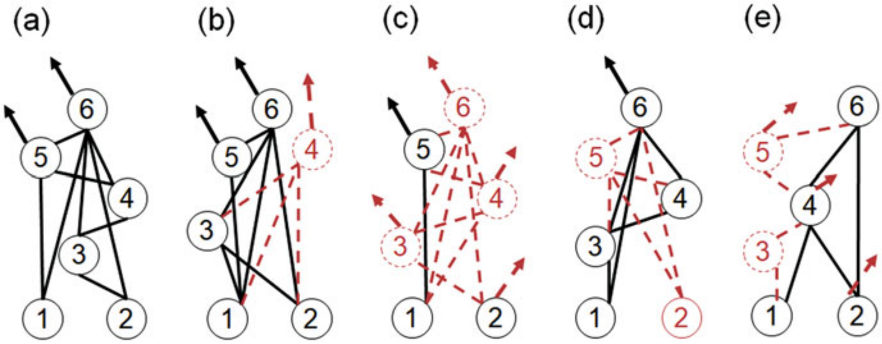


Fig. 18.2 Five simulated examples of multispecies MSY of hypothetical communities consisting of six species by numerical experiments. The edges connecting a pair of species are trophic links, the arrows mean the targets of fishing, and the dashed circles and lines mean the extinct species and their trophic links (Matsuda and Abrams 2006)

exists in the absence of any catch. As a result shown in Fig. 18.2, in the multi-fish species MSY, more than three species became extinct in nearly 30% of the cases, and in only 1% of cases did all six species coexist (Fig. 18.2a). In all cases, the top predator was either eradicated or exploited for fishing. In Fig. 18.2b, the strategy of eradicating species 4 and utilizing 5 and 6 maximized the total yield. Species 2 and 5 are eradicated in (Fig. 18.2d), species 3 and 5 were eradicated in (Fig. 18.2e), and just 2 species survived in (Fig. 18.2c). There was no case that the top predator is kept without harvesting and no case that used all the extant species.

These results, showing that the sustainable use of multispecies fisheries systems does not guarantee the persistence of all species, are very different from the MSY theory of single species. For an extinct species in an exploited community of species, the cause may be direct exploitation of that species or extinction may be caused indirectly by changes in the abundances of interacting species. The solution that maximizes total catch does not necessarily use all fish species, nor does it guarantee the survival of species that do not. Therefore, conservation of biodiversity and maximum sustainable resource use differ in principle. In order to conserve biodiversity, ecological management goals and evaluation criteria should be considered from a perspective independent of sustainable resource use.

Knowledge of community ecology will be widely used in fishery management theory in the future. It has already become an international agreement under the slogan of “ecosystem approach.” However, there are few cases of concrete research. As described in Part II, adaptive population management that takes into account uncertainty and variability may be more popular than ecosystem approaches. The effectiveness of perspectives from community ecology will depend on future examples of management successes using such an approach.

18.4 Maximum Sustainable Ecosystem Services

In 2018, Japan made a major amendment to the Fisheries Act, and fishers with territorial user rights in fisheries became accountable for sustainable use. The revised act also clarified that the catch quota should be set based on MSY. In response, the Japanese Society of Fisheries Science has published an opinion that concerned the use of MSY concepts for two reasons. It is necessary to estimate the relationship between reproduction and stock abundance to determine the catch quota based on MSY, but the uncertainty regarding this is very high. The stock-recruitment relationship is likely to vary greatly between species, and also is likely to vary temporally within a species due to environmental changes. The JSFS suspected that there was no other choice but to adopt adaptive management because there is incomplete knowledge. However, this is not as fundamental an issue as the dependence of classical MSY theory on single species systems.

This second point is essential. The opinion is quoted from the Science Council of Japan (2017), and MSY is a classical concept that focuses only on single species management and does not incorporate interspecific interactions among fish species in the ecosystem. For this reason, they recommended that fisheries resource management should be expanded based on an ecosystem approach that addresses the dynamics of uncertain ecosystems. Ecosystem-based fisheries management must mean not only the use of target resources but also sustainable use of ecosystem services while conserving the biodiversity of the marine ecosystem. We incorporate a precautionary approach and prevent from irreversible damage to the environment, and adaptive management that continuously monitors and evaluates the current status while doing management, and reviews and corrects as necessary. Therefore, it is necessary to maintain the diversity of the ecosystem while cooperating with coastal societies.

Ecosystem services are not only the products of agriculture, forestry, and fisheries obtained from the ecosystem, but also various aspects of "nature's contribution to people" (Pascual et al. 2017). These contributions obtained from the ecosystem include the material cycling, water purification, cultural value obtained from natural products, and a variety of others. As shown in Table 18.1, the food production by fisheries is a part of the provisioning services, the water purification function by bivalves in the tidal flat and the carbon storage of mangrove are the regulating services, and the utilization for tourism is a part of the cultural service.

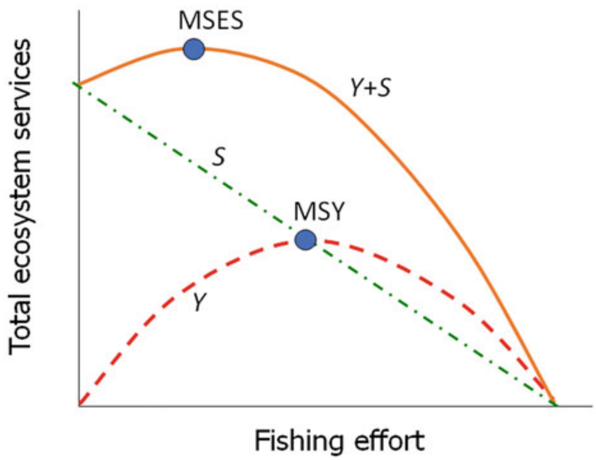
It must not be the MSY that should be maximized sustainably, but the entire set of ecosystem services (Fig. 18.3). Marine and coastal provisioning services are provided primarily by fisheries yield (Y). Among nature's contribution to people (NCP) includes the fishery, but also includes ecosystem services, denoted by S in the figure. Many different aspects of ecosystem health, including many that affect the reproductive rate of fish species, are included in those services. Figure 18.3 shows the conceptual diagram of the relationship between fishing effort and ecosystem services. We simply assumed that the value of NCP is proportional to the abundance of resources. The sum of this Y and S becomes the total value of ecosystem services.

Table 18.1 Summary of monetary values for each service per biome. (Int.$/ha/year, modified from de Groot et al. 2012)

	Marine	Coastal systems	Coral reef	Other	Total
Provisioning services	102	2396	55,724	10,628	68,850
1. Food	93	2384	677	3574	6728
2. Water	0	0	0	3711	3711
3. Raw materials	8	12	21,528	1271	22,819
4. Genetic resource	0	0	33,048	23	33,071
5. Medicinal resources	0	0	0	1905	1905
6. Ornamental resources	0	0	472	146	618
Regulating services	70	26,222	187,688	215,281	429,261
Cultural services	319	300	108,837	10,619	120,075
Total	491	28,917	352,249	236,530	618,187

Note: 1–6 are breakdowns of provisioning services. Regulating services include "habitat services" in the original article. The coral reef is excluded from the coastal area and both are excluded from the ocean

Fig. 18.3 Conceptual diagram of maximum sustainable yield (MSY) and maximum sustainable ecosystem service (MSES, Matsuda et al. 2010)



As shown in Fig. 18.3, the fishing effort (*F*) that maximizes the total ecosystem services will be smaller than the *F* that achieves MSY. Although we do not consider it specifically in Fig. 18.3 some traditional fishing may contribute to the cultural services from tourism in the coastal area.

The practical problem is how to evaluate the value of regulating services. There is also a high degree of uncertainty regarding the value of provisioning services, the majority of which are traded in the market. However, as shown in Table 18.1, the value of coordination services except for the open ocean is much higher than the provisioning services including fisheries. The term MSY is used in articles 61 and 119 of the UN Convention on the Law of the Sea, adopted in 1982. We have long said that the concept of MSY is already outdated (see also Hilborn 2002) and that "maximum sustainable ecosystem services" is a much better concept for the ecosystem approach (Matsuda et al. 2010). Aiming only for maximizing fisheries yield is not an ecosystem approach. Nonetheless, the MSY concept is still being used not only in fisheries scholars but also in documents of the Convention on Biological Diversity and the Intergovernmental Science-Policy Platform for Biodiversity and Ecosystem Services (IPBES).

18.5 Caution to Overconfidence of Adaptive Management

Many may feel that adaptive management, including feedback control, is very useful because it is robust to measurement and process errors. When the concept of MSY was born in the 1930s, the classic MSY was derived from a single species model that ignored interspecific interactions. The classic MSY was also based on a deterministic model that did not consider uncertainties such as measurement errors in resource abundance and that assumed a steady state. Since then, the definition of MSY has evolved into a risk management model that takes into account measurement errors, process errors, and nonequilibrium dynamics.

However, to date, resource management based on the MSY theory is still single species management, not multispecies management, such as target switching strategy as mentioned above (Katsukawa and Matsuda 2003). We defined The MSY from the whole food web, but this does not guarantee species coexistence, as mentioned above (Matsuda and Abrams 2006). Feedback control used in adaptive management is robust to uncertainties and process errors. However, it is vulnerable to species extinction or near-extinction in complex systems, as mentioned below (Matsuda and Abrams 2013).

Optimal control is theoretically possible even in complex systems but is vulnerable to uncertainties such as resource estimation errors. The constant escapement strategy derived from optimal control theory results in overfishing if the stock abundance is overestimated. In addition, fishery profits are only one part of ecosystem services, and ecosystem services should be maximized (Matsuda et al. 2010). The MSY theory does not incorporate the viewpoint of such an ecosystem approach (Matsuda 2012: p. 4), and cannot incorporate it in principle.

In this section, we investigate the effects of species interactions on the robustness of feedback control of the harvesting of prey species. We consider the consequences of feedback control of the fishing effort. If a prey species is exploited, increasing fishing effort decreases predator abundance more than it does the prey abundance. Feedback control of fishing effort may cause the extinction of the predator, even if the prey population is well controlled. Even when fishing effort is controlled by predator density, it is difficult for the fishery and the predator to coexist and the system exhibits complex dynamic behaviors. If the predator and fishery coexist, feedback control of fishing effort converges to a stable equilibrium, a synchronous cycle, or an asynchronous cycle. In the last case, the system undergoes more complex cycling with a longer period than that when the fishing effort is kept constant. These analyses suggest that there is no effective strategy that is robust against measurement errors, process errors, and complex interactions in ecosystem dynamics.

In real fisheries, we rarely know the exact stock abundance or exact surplus productivity. Therefore, we need to incorporate measurement errors into fisheries management theory. In addition, the surplus productivity of fisheries resources often varies from year to year. Therefore, we need to incorporate process errors into population dynamics models (Hilborn and Mangel 1997). Fishermen intuitively realize that fisheries resources increase under favorable environmental conditions even with a high fishing effort, while resources often decrease under poor conditions even with fishing bans. However, this does not mean that fisheries management is unnecessary but rather indicates the necessity of risk management in fisheries (Kawai et al. 2002; Matsuda et al. 1992).

The constant escapement policy is the optimal solution for a population dynamics model with process errors (Reed 1979). This policy is derived from dynamic programming (Iwasa et al. 1984; Mangel and Clark 1988) or optimal control (Kato et al. 2010), but models have shown that it is not robust with respect to measurement errors. Suppose that N_c in (18.2) is set 40% of the carrying capacity (K) and true and estimated abundance (N and $\tilde{N}$, respectively) are 40%K and 80%K, the catch quota will be set $40\%K = \tilde{N} - N_c$, which may result in near-elimination of the population. It is often seen in modern resource management that very roughly 60% of the resources are used and 40% of the resources are conserved. In the past, the estimation error of the resource amount was often about double the size of the resource population. Past stock estimates for Atlantic bluefin tuna have more than doubled between 2014 and 2017 (ICCAT 2017). The long-term fisheries yield is usually high under a constant harvest ratio policy. However, as we rarely know the limit of the constant harvest ratio (Hilborn 2002), the policy is not effective in avoiding overfishing and therefore the harvest ratio depends on the estimate of recent stock biomass in the decision-making process determining the total allowable catch.

More recently, adaptive management has become popular and this is interpreted as a management strategy that is robust to uncertainty and fluctuating stock dynamics (e.g., Walters 1986). Adaptive management consists of two key factors, adaptive learning and feedback control. Adaptive learning is the enforcement of management even without full scientific certainty, the provision of monitoring, and revisions of the management procedures taking into account data from this monitoring. In the case of deer management on Hokkaido Island, Japan, the population size of sika deer (*Cervus nippon yezoensis*) was uncertain but was estimated at 120,000 in 1993 (Matsuda et al. 1999). From catch data until 2000, we found that this was an underestimate, and the population estimate was therefore revised to 200,000 (Matsuda et al. 2002; Yamamura et al. 2008).

The second key concept of adaptive management is feedback control. This means that management measures (e.g., fishing effort) are changed based on the recent measured status of the target population. In the case of deer management, hunting effort depends on the recent index of the relative population size. If trends in relative population size are monitored, the deer population is controllable (Matsuda et al. 1999).

Recently, adaptive management has been recommended in many documents including the Convention on Biological Diversity. However, the robustness of feedback control has not yet been verified using models with interspecific interactions. According to a simple idea of feedback control in fisheries management, the fishing effort depends on the sign and magnitude of the difference between the current stock size and the target size or index (Tanaka 1980).

Matsuda and Abrams (2013) showed that the feedback control of fishing effort on a single stock may result in undesired outcomes, under which mean yields and variability in those yields are suboptimal. They considered the following prey–predator model with the harvesting of prey:

$$\begin{aligned} \frac{dN_1}{dt} &= r_1\left[1-\left(\frac{N_1}{K}\right)^z\right]N_1 - \frac{bN_2N_1}{1+bN_1h} - qE_1N_1, \\ \frac{dN_2}{dt} &= \left[1-r_2-k_2N_2+\frac{mbN_1}{1+bN_1h}\right]N_2, \end{aligned} \tag{18.7}$$

where N_1 and N_2 are the population abundances of the prey and predator, respectively; r_1 is the prey's per capita birth rate; r_2 is the predator's per capita death rate; K is the carrying capacity of the prey; k_2 is the direct density effect on the predator growth rate; z is the positive exponent of the generalized logistic model (Gilpin and Ayala 1973; Pella and Tomlinson 1969); m is the maximum energy conversion rate from prey to predator; b is a positive constant that scales the maximum of the predator's functional response to prey; h is the inverse of the half saturation abundance of the functional response; E_1 is the fishing effort on the prey; and q is catchability (catch per unit effort per unit stock abundance).

Since the stock size may fluctuate, they considered the time-average of yield, which is defined as

$$Y = \frac{1}{T}\int_0^T qE_1N_1(t)dt \tag{18.8}$$

where T is a sufficiently long time interval. They ignore the possible time- and density-dependence of the price p and normalize it to 1 without loss of mathematical generality. We also ignore the economic discounting of future harvests in calculating the yield by expression (18.8).

First, they considered a case of CHR when E_1 is a constant and ignored the direct density effect of the predator ($k_2 = 0$). Due to the nonlinear (type II) functional response ($h > 0$), the stock abundance may fluctuate permanently and the average stock abundance may differ from the equilibrium abundance. They investigated the fishing effort that maximizes the average yield of prey.

If $k_2 = 0$, the equilibrium, denoted by (N_1*, N_2*), is given by

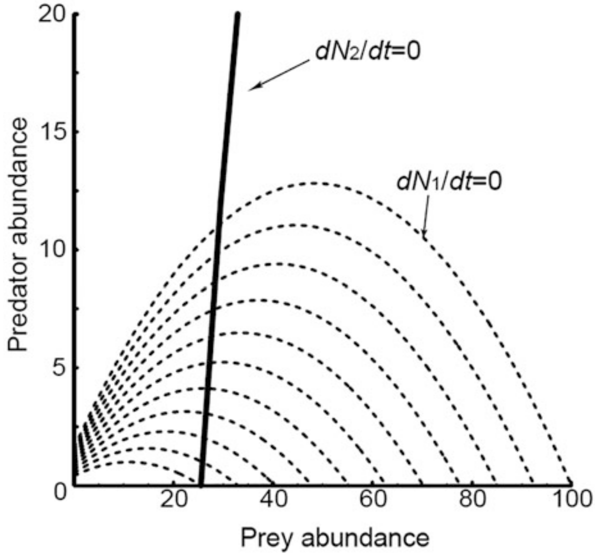


Fig. 18.4 Isoclines of system (18.7). The bold line represents $dN_2/dt = 0$ and the 11 unimodal broken curves from outer to inner represent $dN_1/dt = 0$ for $E_1 = 0.12j$, for $j = 0, 1, 2, \ldots, 10$. Parameter values are: $b = 1$, $r_1 = 1.6$, $r_2 = 2.27$, $h = 0.3$, $K = 100$, $k = 0.003$, $m = 0.77$, $q = 1$, $z = 2$ (Matsuda and Abrams 2013)

$$\left(N_1^*, N_2^*\right) = \left(\frac{r_2}{m - hr_2}, \frac{mr_1}{b(m - hr_2)}\left[1 - \left(\frac{r_2}{b(m - hr_2)}\right)^z\right] - \frac{mqE_1}{b(m - hr_2)}\right) \quad (18.9)$$

In general, the interior equilibrium point is obtained by determining the intersection of two isoclines, $dN_1/dt = 0$ and $dN_2/dt = 0$, as shown in Fig. 18.4. If the predator's direct density effect is 0 or very small, the predator's isocline is close to the vertical line, as shown in Fig. 18.4. The prey's isocline is given by a unimodal curve that diminishes with increasing fishing effort E_1. The equilibrium is given by the intersection of these isoclines. When k_2 is 0 or close to 0, the equilibrium prey abundance N_1* does not vary significantly with changing fishing effort until the predator goes extinct. Thus, the equilibrium yield ($Y^* = qE_1N_1^*$) increases as E_1 increases. In contrast, the equilibrium predator abundance N_2* decreases with increased fishing effort. From the linear stability analysis, if the predator's direct density effect is absent (i.e., $k_2 = 0$), the interior equilibrium is locally stable if $N_2 > N_{2\max}$, where $N_{2\max}$ indicates the N_2* that is maximized on the isocline.

The predator becomes extinct if

$$E_1 \geq E_{1\tau} \equiv \frac{r_1}{q}\left[1 - \left(\frac{r_2}{Kb(m - hr_2)}\right)^z\right] \quad (18.10)$$

If E_1 is between $E_{1\tau}$ and r_1/q, the prey abundance decreases as fishing effort increases. If $E_1 > r_1/q$, the prey goes extinct and the yield becomes 0 at the equilibrium point. In addition, if $N_2^* < N_{2\max}$ and $k_2 = 0$, the equilibrium becomes unstable. Simulations (see also Abrams and Roth, 1994) showed that the system reached a stable limit cycle and the average prey abundances and fishery yield are

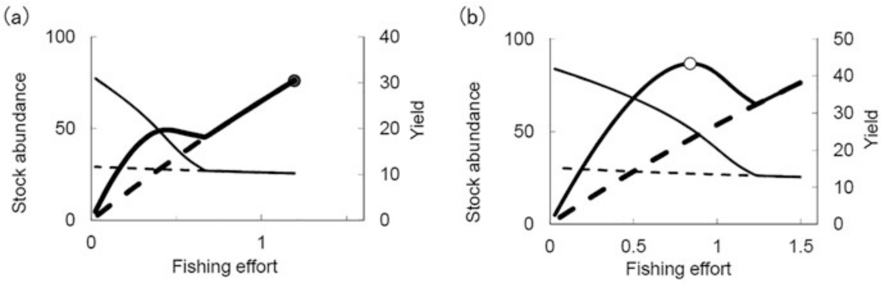


Fig. 18.5 The relationship between Yield and stock abundance versus the fishing effort on the prey. The average (thick line) and equilibrium (broken thick line) yield is humped and is maximized at the filled circle ($E_1 = E_{1\tau}$) in panel (**a**) when the predator goes extinct and at the open circle in panel (**b**). Parameter values are identical to Fig. 18.4 (Matsuda and Abrams 2013)

larger than those at the equilibrium (Fig. 18.4). But we do not know if this is a common outcome for biologically reasonable parameter values. In fact, estimates of parameter values in natural predator–prey systems are quite rare. We should note that the maximum sustainable yield of prey harvesting does not guarantee the persistence of the predator population (Fig. 18.5).

18.6 Feedback Control in Terms of the Resource Density

Matsuda and Abrams (2013) also considered feedback control of fishing effort that varies with resource density. The feedback control of fishing effort in terms of resource density and predator density is described as:

$$\frac{dE_1}{dt} = \begin{cases} U(E_1)(qN_1 - s_1), \text{if } E_1 > 0 \\ \mathrm{Max}[U(E_1)(qN_1 - s_1), 0], \text{if } E_1 = 0 \end{cases} \tag{18.11}$$

where U is a scaling factor describing the rate of adaptive change in the fishing effort; s_1 is the target level of the CPUE “a” in Fig. 18.6. In this case, we ignore measurement errors in the monitoring of resource or predator densities.

Feedback control allows fishing effort to change with its CPUE (qN_1). If the predator’s direct density effect, k_2, is positive, system (18.8) with feedback control (18.11) has an interior equilibrium (N_1*, N_2*, E_1*) given by

$$\left(\frac{s_1}{q}, \frac{bs_1(m - hr_2) - qr_2}{k_2(q + bhs_1)}, \frac{q[qr_2 + bs_1(hr_2 - m)}{k_2(q + bhs_1)^2} - \frac{1}{q}\left[\left(\frac{r_1 s_1}{Kq}\right)^z - r_1\right]\right) \tag{18.12}$$

Linear stability analysis implies that equilibrium (18.10) has either a stable focus or an unstable focus with a stable limit cycle. When the prey–predator system is either unstable without the fishery or includes a direct density effect of the predator,

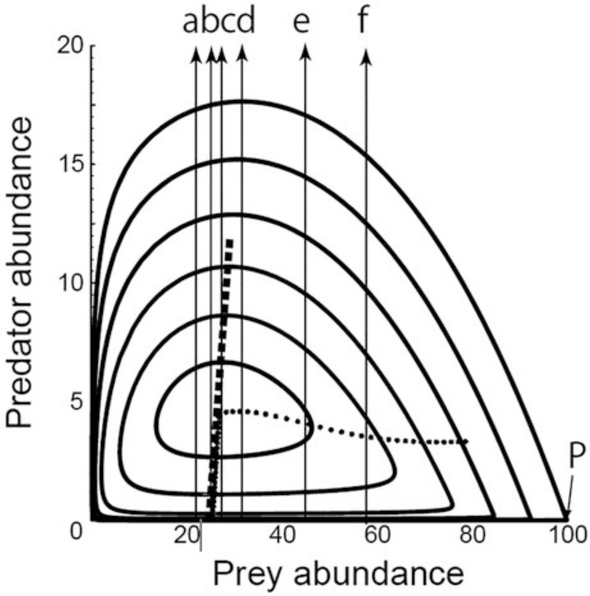


Fig. 18.6 Limit cycles of the prey–predator system (18.8) without the fishery. The broken thick line with an almost vertical slope represents the equilibrium point for $0 < E_1 < 0.12$. Six closed trajectories (broken lines) from outer to inner represent the limit cycles for $E_1 = 0.12j$, for $j = 0, 1,\ldots, 5$. Dots represent the average prey and predator abundances for $E_1 = 0.024j$, for $0 \leq j \leq 50$. If $E_1 > 0.69$, the system has a stable equilibrium. Arrows labeled a–f represent target prey abundance of feedback control (see Fig. 18.4)

three types of outcomes are possible; the extinction of the predator, extinction of the fishery, or coexistence of the predator and fishery. If k_2 is positive, interior equilibrium (18.10) is possible. The isocline of $dE_1/dt = 0$ is given by E_1* at the equilibrium (18.12); this implies a nearly vertical line for the phase plane (N_1, N_2) as shown in. For the existence of interior equilibrium (18.10), the intersection of $dN_1/dt = 0$ and $dN_2/dt = 0$ must exist in the first quadrant of (N_1, N_2). The range of target CPUEs (s_1) for which equilibrium (18.10) exists is usually narrow (Fig. 18.6).

Thereafter, the system consisting of the prey and its fishery has the following stable equilibrium

$$\left(N_1^*, E_1^*\right) = \left(\frac{s_1}{q}, \frac{r_1}{q} - \frac{1}{q}\left(\frac{r_1 s_1}{Kq}\right)\right), \tag{18.13}$$

as shown in E_1's isocline "a" in Fig. 18.6.

In the case of E_1's isocline "b" in Fig. 18.6, the prey, predator, and fishery coexist, and the system approaches a stable equilibrium. Even if equilibrium (18.10) exists but it is unstable, the prey, predator, and fishery coexist and the system approaches a stable limit cycle, as shown in E_1's isocline "c" in Fig. 18.6. Even when the interior equilibrium (18.10) does not exist as shown in E_1's isocline d–f in Fig. 18.6, the prey, predator, and fishery can coexist. As is well known in two−predator−one−prey system, these three species may coexist if the population fluctuates far from its average abundance (Abrams and Holt 2002: Armstrong and McGehee 1980). If $k_2 = 0$, no interior equilibrium like (18.10) exists. System (18.8) is mathematically equivalent to the system analyzed by Armstrong and McGehee (1980) if U in (18.11)

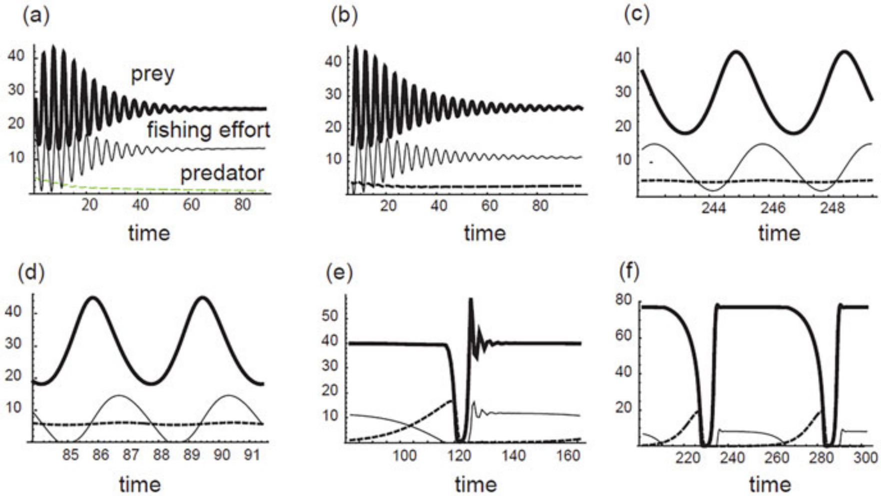


Fig. 18.7 The dynamics of prey abundance (thin lines), predator abundance (thick lines), and fishing effort (broken lines) with a variety of target CPUEs (s_1). Panels **a–f** indicate the trajectories when s is set to **a–f** shown in Fig. 18.6, respectively. (**a**) The fishery goes extinct ($E_1 \rightarrow 0$); (**b**) the prey (N_1), predator (N_2), and fishery (E_1) coexist at a stable equilibrium; (**c**, **d**) N_1, N_2, and E_1 converge to a synchronous cycle; (**e**, **f**) These dynamics form an asynchronous cycle with a more complex fluctuation (Matsuda and Abrams 2013)

is proportional to E_1 ($dE_1/dt = UE_1(qN_1 - s_1)$), but here a constant U is assumed. Therefore, the predator and fishery may coexist when prey abundance fluctuates permanently.

If the equilibrium without fishery is unstable and the system undergoes a stable limit cycle, the CPUE may be larger than the target level (s_1) at some time periods during the cycle. If $\max[N_1(t)] > s_1/q$, the fishing effort is positive for at least some time periods, as shown in E_1's isoclines "d–f" in Fig. 18.6.

If the original prey–predator system (18.8) is without a fishery and has no density-dependence ($E_1 = k_2 = 0$), coexistence between a predator and a fishery that shares a common resource is impossible with feedback control.

Figure 18.7 shows the trajectories of system (18.8) with (18.11), with a variety of target CPUEs (s_1). If the predator and fishery coexist, feedback control of the fishing effort converges to either a stable equilibrium, a synchronous cycle, or an asynchronous cycle. In the last case, the system has a more complex cycle with a longer period than that which occurs when the fishing effort is kept constant. In the course of the asynchronous cycles in Fig. 18.7e, the predator abundance $N_2(t)$ gradually increases until $t = 120$. When $t = 120$, the prey abundance and fishing effort begin to fluctuate, the predator abundance soon drops dramatically, and the prey abundance and fishing effort then resume their pattern of gradual change.

These results suggest that feedback control in fisheries can avoid causing predator extinction and result in fisheries persistence if the target prey abundance is slightly larger than the equilibrium abundance when the fishery is absent. However, if the target prey abundance is smaller than that without fishery pressure, the fishery causes the extinction of the predator. If the target prey abundance is too much larger, the fishery causes complex fluctuations in the prey, predator, and fishing effort. While the prey is decreasing, even a zero catch does not prevent the decline.

The general message from this section is that feedback fisheries management may cause the extinction of species that utilize the target of the fishery, or may cause the extinction of the fishery itself. Feedback control that only responds to the abundance of the harvested species risks causing extinction when that species has a specialist predator. Managing the prey by responding to the abundance of a specialist predator may be better.

Adaptive population management is clearly defined and is described by mathematical forms, e.g., the Revised Management Procedure of the International Whaling Commission (Tanaka 1980). Bayesian statistics and population dynamics models with feedback control of human activities (e.g., harvest rate) play major roles in adaptive population management. Adaptive management is also popular in ecosystem-based management or ecosystem approaches. However, ecosystem approaches in practice rarely include Bayesian statistics and feedback control of management measures. Ecosystem models are often incorporated into ecosystem-based management, but such models seldom incorporate uncertainties about measured population sizes or the rules governing population dynamics. A gap also exists between adaptive population management and ecosystem-based adaptive management in the kind of models that are most often used. The latter usually uses ecosystem models, but the concept of adaptive management is often used just as a qualitative "plan-do-check-act" (PDCA) cycle or "learning-by-doing."

Adaptive management is not a description of a decision-making process or a process of policy change; instead, it is a design for how to control the objective system to be in or near to a desirable state. The expectation is that feedback control on human activities in terms of the state of the target should be desirable. For example, one might expect that feedback control on the harvest rate of a target fisheries resource in terms of the stock abundance of that resource would avoid overfishing. In this paper, we showed that this outcome is far from always being true.

Feedback control is robust against measurement errors and process errors (Walters 1986). It is effective for controlling a dynamic system consisting of one variable. In addition, even if we do not know the details of surplus production (e.g., the intrinsic growth rate and carrying capacity), feedback control is effective in avoiding stock collapse and can enhance sustainable fisheries. When we analyzed the case of feedback control in terms of predator density, we implicitly used knowledge of prey–predator dynamics. If we do not know the true model of this dynamic, we cannot design effective feedback control strategies. Feedback control may not be effective for controlling a dynamic system consisting of multiple variables like prey–predator systems. The message arising from this simple system is likely to apply to most larger food webs in which more mechanisms are producing dynamic instability. The work of Yodzis (1998) suggests that relatively small errors in estimating population growth parameters can make large differences in the effects of fishing a single species in a large food web, which is called "indeterminacy".

We do not know any effective strategy or management tool that is robust against measurement errors, process errors, and complex interactions in ecosystem dynamics. We need information about key elements of the processes governing ecosystem dynamics to derive a policy that is robust against these uncertainties. Future calls for adaptive management and ecosystem management should keep these limitations in mind.

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