INSTITUTIONS AND THE ENVIRONMENT

Elinor Ostrom

Scholars have tended to recommend 'optimal' solutions for coping with open-access problems related to common-pool resources such as fisheries, forests and water systems. Examples exist of both successful and unsuccessful efforts to rely on private property, government property and community property. After briefly reviewing how the often-recommended solutions have worked in the field, I suggest that institutional theorists move from touting simple, optimal solutions to analysing adaptive, multi-level governance as related to complex, evolving resource systems.

Introduction

The first thing that an institutional theorist wants to do when given an assignment to write a chapter on 'Institutions and the Environment' is to clarify how these concepts will be defined. In everyday parlance, the terms 'institutions' and 'environment' are used casually and refer to many things. Sometimes people refer to a local prison as an institution, or to a broad practice within a society, such as 'the institution of marriage'. The 'environment' can be used to refer to the immediate area surrounding a particular setting or to the global atmosphere. Fortunately, over time, ever clearer and more useful definitions for institutions, for the diverse forms of 'the environment', and as well as for the linked levels of interaction, are being developed and used by researchers – particularly those interested in how institutions enhance or adversely affect multiple objects and processes related to ecological systems (see Aoki, 2001; North, 2005; Ostrom *et al.*, 2007).

In this chapter, the term *institutions* refers to the *rules* that humans use when interacting within a wide variety of repetitive and structured situations at multiple levels of analysis (North, 2005; Ostrom, 2005). Individuals who regularly interact use rules (or the absence of rules) designated by government authorities as relevant for situations of a particular type. They may also develop and enforce their own rules. Individuals interacting within a particular rule-structured situation linked to a specific environment may also adopt norms regarding their behaviour given the others who are involved and their actions over time. In light of the rules, and shared norms when relevant, individuals adopt strategies leading to consequences for themselves and for others (Crawford and Ostrom, 1995). As individuals learn more about the outcome of their own and others' actions within a particular situation, they may change norms and strategies leading to better or worse outcomes for themselves and the relevant environment.

Many environmental goods are common-pool resources, which will be the focus for this article. *Common-pool resources* include resources that are sufficiently large that excluding potential beneficiaries from using them for consumptive or non-consumptive purposes is non-trivial. Each individual consumptive use (for example, harvesting a truckload of forest products or withdrawing water from an irrigation system) reduces the resource units that are available to others (Ostrom and Ostrom, 1977; Ostrom *et al.*, 1994). Without effective institutions to limit who can use diverse harvesting practices, highly valued, common-pool resources are overharvested and destroyed (FAO, 2005; Mullon *et al.*, 2005; Myers and Worm, 2003).

Modelling the open-access problem

Developing formal models has been an important tool for institutional theorists for analysing why common-pool resources are overharvested and what might be done to avert their destruction. One of the earliest, most powerful and long-lasting models of a common-pool resource is the static model of a fishery published by Scott Gordon in 1954. In

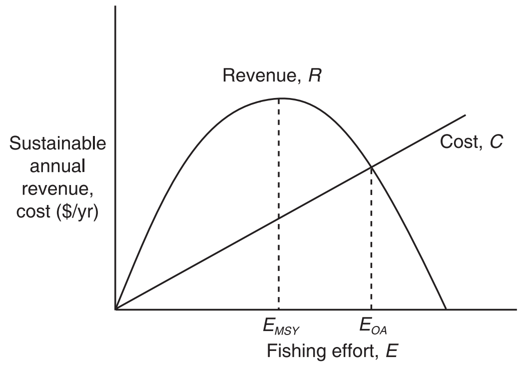

Figure 1: The Gordon model of fishery bioeconomics
Source: Clark (2006, p. 11).

an open-access fishery, Gordon (and many other scholars who have drawn extensively on his work) posited that each fisher would invest effort in harvesting until they reached a bionomic equilibrium, E_{OA}, where *total* revenue equals *total* cost. The bionomic equilibrium (E_{OA} in Figure 1) generates a high level of rent dissipation. If fishers were to fish at a maximum sustainable yield E_{MSY}, substantial economic gains would be achieved. The problem is that each fisher still makes money at E_{MSY} and other fishers would want to enter the fishery. As long as the fishery is open to any fisher who wants to earn money, the bionomic equilibrium will persist rather than the maximum sustainable yield.

This static model has repeatedly been used to show why common-pool resources that generate highly valued resource units will be overharvested when no effective rules limit entry or withdrawals. The power of the Gordon model comes from the clarity of its representation of why unregulated common-pool resources are overharvested. On the other hand, its simplicity is also a weakness when used for designing new institutions to overcome economic incentives to overharvest. As Colin Clark (2006, p. 15) reflects, the static, 'stick-figure' model is too simplistic for analysts to apply it as if it adequately described all common-pool resources. The presumption of many analysts has been that all one has to do is to impose rules so that harvesters face different incentives and withdraw at a maximum sustained yield.

The Gordon model has been used as the underlying theoretical model for the design of a series of laboratory experiments on behaviour related to common-pool resources. The predictions of the Gordon model regarding overharvesting are supported when subjects make anonymous decisions and cannot communicate with one another (Ostrom *et al.*, 1994). Given an opportunity to use 'cheap talk' where promises to one another are not externally enforced, subjects adopt norms and strategies that enable them to achieve higher returns (Ostrom and Walker, 1991). Further, when given an option to covenant with one another regarding whether to create their own sanctioning system to be used against non-cooperators, subjects who develop their own sanctioning system approached the achievement of optimal outcomes (Ostrom *et al.*, 1992).

The predictions of overharvesting from an *open access* common-pool resource are thus supported in the field and in the lab. Designing institutions so that harvesters are motivated to harvest at a sustainable level is the problem that needs to be addressed with knowledge about particular resources rather than just accepting the 'stick figure' as representing all one needs to know about open access resources. Unfortunately, too many reforms are presented as not much more than 'stick figures', and scholars, policy-makers and communities have had to learn the hard way about finding rules that match the complex ecologies that are involved in diverse common-pool resources.

Recommending optimal institutions

The widespread acceptance of the Gordon model has led policy analysts to recommend three idealised institutions to induce individual users to engage in sustainable harvesting practices. Some of the rules recommended as 'optimal' are private property (Demsetz, 1967; Raymond, 2003), government ownership (Lovejoy, 2006; Terborgh, 1999, 2000), or community control (Vermillion and Sagardoy, 1999). Multiple examples exist where moving to government ownership, private property or community control of a common-pool resource has worked to help users achieve more efficient short-term results and potentially to sustain the resource over the long term. The particular arrangements that have proven to be effective, however, differ radically from one another and from the simple policy recommendations made by scholars recommending 'optimal' solutions (Rose, 2002; Tietenberg, 2002).

Private property and common-pool resources

In southern California after World War II, for example, groundwater producers used the California courts as an arena in which to determine who had rights to pump how much water per year. The courts also established a 'watermaster' to determine the factual information initially needed to determine rights and then to monitor the adherence of water producers to the agreements (Blomquist, 1992). In the groundwater basins that were adjudicated and rights allocated, markets for water rights emerged rapidly. Furthermore, water rights tended to be sold or leased by those who had lower marginal productivity to those with higher marginal productivity – such as water companies who needed rights to pump water to meet peak demands – and by rights holders who were exiting the resource (either by moving away, ceasing or changing their business) to users who wished to expand their access to local water sources.

After half a decade, times have changed in regard to the population of the region, local water sources and water availability in several linked aqueducts. The continuing jurisdiction of the California court system has enabled water producers to adjust the rules they had earlier negotiated to cope with disturbance and changing conditions (Blomquist and Ostrom, 2008; Steed and Blomquist, 2006). In some years, producers were authorised to take more than their assigned rights so long as they then curtailed their water production at a later time (similar to receiving a monetary loan from the bank that has to be paid back). And, in some cases, producers were authorised to take less than their assigned shares and 'bank' or store water for future withdrawal. Furthermore, the water producers have experimented with a diversity of other institutions, such as the creation of special districts to levy a substantial tax on pumped water, to pay for basin replenishment as well as monitoring and reporting on basin conditions. Thus, while privatising rights was a crucial step in reducing continued overharvesting of groundwater in Los Angeles, it was only one of a complex series of institutional changes and adaptations over time.

In relationship to fisheries, individual transferable quota (ITQ) systems are frequently recommended as the 'optimal' strategy (Raymond, 2003; Scott, 1988). Notable cases exist where establishing an ITQ system has averted a collapse of a fishery, but few of the 'successes' were immediate. All took some time adjusting various aspects after a national government agency first designed an ITQ system. Most of the successes have evolved into more complex systems relying on multiple institutional arrangements rather than being simple ITQ systems.

The British Columbia trawl fishery for groundfish, for example, had been heavily utilised since World War II (Grafton *et al.*, 2006). Early efforts to control overfishing by governmental policies included: restricting the number of fishing vehicles and the equipment that could be used, the assignment of total allowable catch (TAC) quotas, and the assignment of fishing trip quotas. Massive overharvesting led to the closing of the fishery in 1995. Within a few years, the fishery was reopened with new regulations including an annual ITQ system granted by the Federal Minister of Fisheries for each species (Clark, 2006, pp. 238–240). Thus, fishers do not 'own' the quota assigned, but some trading is allowed, and no ITQs have been taken away from assigned trawlers. In addition, all catches are recorded by onboard observers to avoid earlier problems of underreporting. Clark (2006, p. 239) observed that the ITQ system has led to profound changes:

> 'First, catch data are now reliable, allowing the scientists to perform believable TAC estimates. [This is the result of the observer programme, not of the ITQ system itself, although the latter no doubt implies a degree of acceptance and support of the observer programme.]
>
> 'Second, a decrease in fleet capacity has occurred, as both small and large vessels have sold their quotas and withdrawn from the fishery . . .
>
> 'In terms of resource conservation, discards are not only accurately quantified, but have also been significantly reduced because of the ITQ-generated economic incentives against catching unwanted species.'

Thus, the ITQ system has had a positive impact on the fishery, but an effective monitoring system was also an essential aspect of the success. The importance of realistic provisions for monitoring the conformance of resource users to a property rights allocation is rarely mentioned when scholars recommend ITQ systems.

While smaller-scale inshore fishing had long existed in New Zealand's inshore waters (Johnson and Haworth, 2004), large-scale, deepwater, commercial fisheries developed later in New Zealand than in British Columbia due to heavy fishing by foreign fleets in waters surrounding New Zealand. New Zealand declared its 200-mile Exclusive Economic Zone in 1983 and the government immediately began to offer incentives to domestic, commercial fishers to encourage them to replace the foreign fleets that had been overfishing their waters. In 1986, New Zealand became one of the first countries to adopt a market-based fishery regulation when it adopted a quota management system (QMS) and allocated ITQs to a subset of domestic fisheries (Annala, 1996). The government also removed the subsidies it had established just a few years earlier.

New Zealand authorities had to make still further adjustments as they discovered that the biological models underlying the initial allocation of permanent allocation of fixed quotas needed to be adjusted over time in light of further evidence. After considerable renegotiation, commercial fishers received a revised ITQ in 1990 based on a proportion of the total catch assigned annually (Yandle and Dewees, 2003). Commercial fishers also demanded a greater role in determining quotas and other fishery policies resulting in the Fisheries Amendment Act being passed in 1999, which recognised commercial stakeholder organisations (CSOs) as 'approved service delivery organizations' that may compete for contracts to provide scientific research and other fisheries services. Furthermore, CSOs are essentially the recognised voice of the industry for the fishery they represent. In essence, the original ITQ system evolved into a 'co-management' system but one in which some major interests – such as customary Maori interests, recreational fisheries and environmental groups – are not recognised (Yandle, 2003). Furthermore, Yandle (2007) has identified some of the 'mismatches' involved in the relationships among the property rights assigned to different groups along both temporal and spatial dimensions (Cash *et al.*, 2006). In regard to spatial mismatches, Yandle (2007) identifies overlapping or poorly defined boundaries such as those among:

> 'customary Maori fishing, aquaculture, marine reserves, and commercial fishing [that] create political or physical competition for access to marine resources, as well as frustration within the commercial fishing community, which perceives that its broad, but not exclusive, spatial rights are eaten away by the smaller but more exclusively defined spatial rights of interests such as marine reserves, customary Maori fishing, and aquaculture.'

The multiple problems of reconciling diverse interests with multiple types of property rights in a spatial domain have led to rapid legislative changes that tend to make the overall system more fragile (Yandle, 2008).

One of the most famous (or infamous, depending on the view of the reader) ITQ systems was gradually introduced in Iceland after multiple crises in Icelandic fishery stocks (Arnason, 1993). After experimenting with diverse TAC systems and considerable controversy, a uniform ITQ system was adopted in 1990 for all relevant fisheries (Eggertsson, 2005). Similar to the evolved New Zealand ITQ system, quotas are not to fixed quantities but rather to a 'share' of the annual authorised catch level set by the Icelandic government based on the recommendation of fishery experts. The Iceland ITQ system appears to have averted the collapse of many valuable species for the Iceland fishery. It has been less successful in restoring the Icelandic cod stocks, which have suffered dramatic losses throughout the North Atlantic region (Finlayson and McCay, 1998).

In his analysis of the long and conflict-ridden road to the Icelandic ITQ system, Eggertsson (2004) reflects that introducing major institutional changes is a 'subtle art' rather than a simple use of a 'one-size-fits-all' formula. Eggertsson criticises the work of fellow economists who have created a sense of 'false optimism' about how to manage complex fisheries. Simply designing a system in a top-down fashion and imposing it on the harvesters does not turn out to be as successful or adaptive as working with the users of a common-pool resource over time to develop a system that is well matched to the ecological system as well as to the practices, norms and long-term economic welfare of the participants.

Government property and common-pool resources

For some scholars, public ownership of land is the *only* way to achieve sustained conservation over time (Lovejoy, 2006; Terborgh, 1999). This has led to proposals for creating a system of government-protected areas across the world (Ghimire and Pimbert, 1997). Currently, more than 100,000 protected areas already exist and include approximately 10% of the forested areas in the world (Barber *et al.*, 2004). While considerable enthusiasm exists for creating protected areas, their performance has varied substantially.

Some positive evaluations of the effectiveness of protected areas rely on qualitative ratings by government officials and park managers at multiple sites rather than independent studies (Bruner *et al.*, 2001; Ervin, 2003). While it is important to learn what officials think about their progress, full reliance on self-assessments may introduce serious biases in the analysis (Hockings, 2003; Nepstad *et al.*, 2006). A study of forest conditions evaluated by an independent forester or ecologist for 76 government-owned protected parks as contrasted to 87 forests owned under a diversity of arrangements (private, community, government) did not find *any* statistical difference in the forest conditions between protected areas and all others (Hayes, 2006; see also Gibson *et al.*, 2005).

A large study conducted by the World Wildlife Fund (WWF) included over 200 protected areas in 27 countries. The WWF found that many protected areas lacked key financial and human resources, a sound legal basis, and did not have effective control over their boundaries (WWF, 2004). Owing to these conditions, extensive conflicts among park residents, park personnel and with local communities that surround many protected areas are frequently reported as well as illegal harvesting (Wells and Brandon, 1992). Nepstad *et al.* (2006) broadened the debate by examining several different tenure arrangements within protected areas including extractive reserves, indigenous territories and national forests in Brazil. Under conditions of intense colonisation pressures, they found that strictly protected areas are more vulnerable to deforestation and fire than indigenous reserves. These and other studies indicate the need to shift away from the presumption that creating government-owned parks and reserves is the *only* way to protect forests and biodiversity.

Carefully controlled analyses over time of remotely sensed images of deforestation levels in national parks located in the *same* country have found that some are well protected and others not. Ostrom and Nagendra (2006) provide strong evidence that the Mahananda Wildlife Sanctuary in West Bengal, India, has successfully prevented deforestation, but this success involves high administrative costs and considerable conflict with the local population. On the other hand, the Tadoba Andhari Tiger Reserve in Maharashtra, with only a modest budget, is not able to control entry into the forest and the loss of forested land is substantial. Forests within Tikal National Park in the Mayan Biosphere Reserve in Guatemala – well-financed through fees collected from tourists – are in excellent condition (Dietz *et al.*, 2006). At the same time, nearby national parks – Laguna del Tigre National Park and the Sierra del Lacondon National Park – even though they are the same ecological zone and under the same institutional structure, are ravaged by illegal harvesting.

Community property and common-pool resources

While some scholars have been overly enthusiastic about the performance of diverse kinds of community ownership or involvement as a solution to overharvesting of common-pool resources (Western and Wright, 1994), strong involvement of a community is no more a panacea than private or governmental ownership (Campbell *et al.*, 2001; Meinzen-Dick, 2007; Nagendra, 2007). Empirical studies of common-pool resources under community control have shown that benefits are sometimes distributed in an unequal fashion among community members (Oyono *et al.*, 2005; Platteau, 2004) leading in some cases to the exclusion of the poorest members of a community (Malla, 2000).

Little evidence exists that *simply* turning common-pool resources over to local users will avoid overharvesting. Some communities manage their fisheries or forests better than others (Acheson, 2003; Andersson, 2004; Gibson *et al.*, 2000). While strong evidence exists that local communities are capable of creating robust local institutions for governing local resources sustainably (Bray and Klepeis, 2005; NRC, 2002; Ostrom, 1990, 2005), some analysts have gone overboard and proposed community-based conservation as another cure-all (Berkes, 2007). This has led some donor-funded efforts to turn control over to local residents with a simple blueprint approach (Pritchett and Woolcock, 2004), leading to little community involvement and enabling local 'elite capture' of benefits. Moreover, total 'turnover' ignores the necessity of managing common-pool resources at multiple levels, with vertical and horizontal interplay among institutions.

Community management in a variety of forms – direct ownership, government concessions, or other long-term co-management arrangements – has the capacity to be as effective or, under certain conditions, more effective than government ownership (Bray *et al.*, 2005). The debate over the effectiveness of diverse institutions needs to be extended to a larger landscape of tenure regimes than just community ownership. Various forms of co-management do assign substantial management responsibilities and access to resources in and around a resource, and a wide variety of community management types, from full ownership to community-rights concessions on public lands to private management, can be effective if they are well tailored to the particular attributes of a resource and the larger and smaller resources to which it is linked. Simple solutions do not exist for managing complex ecologies (Campbell *et al.*, 2006; McPeak *et al.*, 2006).

From optimal solutions to adaptive multi-level governance

A key finding from decades of in-depth studies of institutions and the environment is that the same rules that work well in one setting are part of failed systems elsewhere! There are no 'optimal' rules that can be applied to all fisheries, all forests, or all water systems (Grafton, 2000; Ostrom, 2007). We simply must stop relying on stick-figure models alone and proposing 'one-size-fits-all' solutions, given that these solutions have themselves generated tragedies when widely applied rather than solved them.

Instead of presenting stick-figure models of resource systems, institutional theorists need to recognise what ecologists recognised long ago: the complexity of what we study and the necessity of recognising the non-linear, self-organising and dynamic aspects as well as the multiple objectives and the spatial and temporal scales involved. As the distinguished ecologist Simon Levin (1999, p. 2) has summarised:

> 'That is, ecosystems are complex, adaptive systems and hence, are characterized by historical dependency, complex dynamics, and multiple basins of attraction. The management of such systems presents fundamental challenges, made especially difficult by the fact that the putative controllers (humans) are essential parts of the system and, hence, essential parts of the problem . . .
>
> 'There are a number of lessons that emerge from this study and guide it. Most important is the importance of experimentation, learning and adaptation.'

Institutional economists need to recognise that deriving a simple, beautiful mathematical model is not the only goal of our analysis. Adopting more complex approaches – including flow charts, simulations, dynamic systems analysis and the specification of multiple factors – is not a sign of failure when the systems being analysed are fundamentally complex and multi-level (Wilson, 2006; Wilson *et al.*, 2007). Models are powerful tools and we need to develop them so that they can be used to capture more complex phenomena (Costanza *et al.*, 2001). We cause harm, however, by recommending one-size-fits-all institutional prescriptions based on overly simplified models of resources to solve problems of overharvesting.

Thinking about policy recommendations

In earlier efforts to analyse which rules worked best related to fisheries, irrigation systems and forests, we found a simply gigantic number of individual rules that were used in the field (Ostrom, 2005; Schlager, 1994; Tang, 1994). Focusing on boundary rules that define who can gain access to enter and harvest from a resource, we have identified three broad classes:

- 'Residency or Membership Rules' that specify residency or membership requirements.
- 'Personal Characteristic Rules' that require ascribed or acquired personal attributes (e.g. age, gender, education, skill test, etc.).
- 'Relationship to a Resource Rules' that specify conditions of use depending on the relationship of a user with the resources (e.g. length and continuity of use, ownership of land or other asset, acquisition of licence, etc.).

We have found empirical examples of four types of Residency or Membership Rules, nine types of Personal Characteristic Rules, and 13 Relationship to a Resource Rules. Some specific boundary rules specify more than one category (e.g. a user must be over 21, have taken a skill test, and use a specific type of technology to be an authorised user of a particular resource) (see Ostrom, 2005, Ch. 8).

It is important to note that repeated studies have not yet found *specific* rules that have a statistically positive relationship to performance in a large number of common-pool resources (Dietz *et al.*, 2006; Gibson *et al.*, 2000; NRC, 2002). On the other hand, the absence of *any* boundary rule or *any* monitoring effort to ensure that a well-defined set of authorised users are following the rules related to timing, technology and quantity of harvesting *is* consistently associated with poor performance (Ostrom and Nagendra, 2006; Ostrom *et al.*, 1994).

After reading and coding hundreds of cases that described both successful and unsuccessful private, government and community property arrangements, without finding a clear set of *specific* rules associated with long-term sustainability, I derived a set of design principles to characterise those cases of local, common-pool resources that had survived long periods of time (Ostrom, 1990). The predictive power of these design principles in helping to distinguish successful from unsuccessful cases has now been supported by multiple studies (Dayton-Johnson, 2000; Marshall, 2005; Sarker and Itoh, 2001; Trawick, 2001; Weinstein, 2000).

To apply what we have learned to policy, we can translate the design principles into a set of questions that those involved in designing and adapting institutional arrangements for a particular resource system would need to address. Basically, any institutional arrangement for regulating a common-pool resource to achieve multiple objectives needs to help harvesters and officials address the following questions in a way that is understood by those involved and considered legitimate given the characteristics of the resource, the community involved and the larger economic and political domains:

- Who is allowed to harvest which kinds of resource units?
- What will be the timing, quantity, location and technology used for harvesting?
- Who is obligated to contribute resources to maintain the resource system itself?
- How are harvesting and maintenance activities to be monitored and enforced?
- How are conflicts over harvesting and maintenance to be resolved?
- How will cross-scale linkages be dealt with on a regular basis?
- How will risks of the unknown be taken into consideration?

- How will the rules affecting the above be changed over time with changes in the performance of the resource system, the strategies of participants and external opportunities and constraints?

Instead of presuming that one can design an optimal system in advance and then make it work, we must think about ways to analyse the structure of common-pool resources, how these change over time, and adopt a multi-level, experimental approach rather than a top-down approach to the design of effective institutions.

Experimenting with rule changes

We need to understand the institutional design processes involving an effort to tinker with a large number of component parts (see Jacob, 1977). Those who tinker with any tools – including rules – are trying to find combinations that work together more effectively than other combinations in a particular setting. Policy changes are experiments based on more or less informed expectations about potential outcomes and the distribution of these outcomes for participants across time and space (Campbell, 1969, 1975). Whenever individuals agree to add a rule, change a rule, or adopt someone else's proposed rule set, they are conducting a policy experiment. Moreover, the complexity of the ever-changing biophysical world combined with the complexity of rule systems means that any proposed rule change faces a non-trivial probability of error.

When rules related to common-pool resources are made by a single governing authority for an entire nation, policy-makers have to experiment simultaneously with *all* of the common-pool resources within their jurisdiction with each policy change. For very small countries with similar ecosystems, this may not be a problem. For large countries, however, rules that are appropriate in one region are rarely effective in another. And, once a change has been made and implemented, further changes will not be made rapidly. The process of experimentation will usually be slow, and information about results may be contradictory and difficult to interpret. A policy change that is based on erroneous data about one key structural variable or a false assumption about how actors will react, can lead to a major disaster (see Berkes, 2007; Brock and Carpenter, 2007). Further, as Dixit (2004) has shown, arbitrary policy changes and tax laws made by a highly centralised governance regime may result in substantial rent seeking and graft.

In any design process where there is a substantial probability of error, having redundant teams of designers has repeatedly been shown to have considerable advantage (see Bendor, 1985; Landau, 1969, 1973; Page, 2007). Given the logic of combinatorics, it is impossible to conduct a *complete* analysis of the expected performance of all of the potential rule changes that could be made to change the incentives of resource users. Instead of developing models that generate optimal outcomes, we need to understand what level of redundancy, overlap and autonomy help to adapt rules that work for particular resources under specific social–economic conditions. And, then, we need to focus on how to enhance the robustness of these institutions to diverse disturbances that will 'hit' them over time (Anderies *et al.*, 2007; Janssen *et al.*, 2007).

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Elinor Ostrom is co-director, Workshop in Political Theory and Policy Analysis, Indiana University, and founding director, Center for the Study of Institutional Diversity, Arizona State University, USA (ostrom@indiana.edu).