KEYNOTE LECTURE

why do we need to protect institutional diversity? *

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Abstract

In past decades, scholars have tended to recommend 'optimal' solutions for coping with common-pool resources. Examples exist of *both* successful and unsuccessful efforts to establish government property, private property, or community property. The absence of any property rights – open access – related to valuable resources is associated with overuse. The resource institutions that research has documented as working well in the field differ substantially in their detailed design but can usually be characterised as adaptive, multilevel governance systems related to complex, evolving resource systems. We need to overcome the tendency to recommend panaceas and encourage, instead, considerable experimentation at multiple levels to reduce the threats of massive collapses of valuable resources.

Keywords institutional analysis; common-pool resources; rules; government property; private property; community property

THE PROBLEM OF INSTITUTIONAL MONOCROPPING

Peter Evans (2004: 31–32) cogently pointed to a major problem of contemporary policy analysis when he reflected: 'Currently, the dominant method of trying to build institutions that will promote development is to impose uniform institutional blueprints on the countries of the global South – a process that I call "institutional monocropping"'.¹ Even more problematic than the initial

problem of having the wrong institutions widely imposed is the 'lock in' that can occur when powerful individuals gain advantage from such institutions leading to major problems of path dependence. The poor and helpless are the ones who pay the big costs.

Lant Pritchett and Michael Woolcock (2004) developed a complementary analysis to that of Peter Evans. They puzzle over the problem of new solutions when the dominant Weberian paradigm was *the* solution used by development agencies, and now it is *the* problem

confronting anyone concerned about development. They graphically describe the systematic failure of development agencies to improve any services to rural areas including those related to irrigation. 'Donor activity often amounts to sending "experts" who operate institutions in "Denmark" to design institutions in "Djibouti". At best this would be like sending a cab driver to design a car' (Pritchett and Woolcock, 2004: 199).

How can we get out of the kind of institutional monocropping that currently dominates much of social science thinking as well as that of development agencies? (see Gibson *et al*, 2005). There is obviously not one way to solve this problem! As academics, we can help by being willing to develop more complex theories for explaining the behaviour of humans in widely divergent settings. We do not need to be complex, just to be complex. But we need to get over our simplicity hang-ups. Yes, following a scientific approach, we want our analysis to be as simple as feasible given the problems we are interested in studying. But 'keeping it simple' is a stupid response, when what we are studying are complex social-ecological systems that are nested into many levels. Obviously, our theories will always be simpler than the world we study. Otherwise, we are trying to reproduce the world itself rather than theories about the world that can be tested.

When a single governing authority makes decisions about rules for an entire region, policymakers have to make plans for *all* of the territory within a jurisdiction, with each policy change. And, once an initial policy 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. Thus, a policy that is based on erroneous data about one key structural variable, or one false assumption about how actors will react, can lead to a very

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large disaster (see Wilson, 2006). In any design process where there is substantial probability of error, having redundant teams of designers has repeatedly been shown to have considerable advantage (see Landau, 1969, 1973; Bendor, 1985).

For example, let us imagine a series of inshore fisheries located along the coast of a region and posit that every policy innovation has a probability of failure of 1/10. If the region were regulated by a single governing agency, one out of ten policy changes would be failures for the entire region. If designing rules were delegated to three genuinely independent authorities, each of these authorities would still face a failure rate of one out of ten. The probability that a failure would simultaneously occur along the entire coast, however, would be reduced from 1/10 to 1/10³ or 1/1000. On a coast with many relatively separable inshore fisheries that are governed by local authorities, the likelihood of a *coastal-wide* failure is reduced still more. Of course, the failure rate for such design tasks can itself not be known, but the positive effect of parallel, redundant design teams each trying to find the best combination of rules does not depend on any particular error rate. The important point is: If the systems are relatively separable, allocating responsibility for experimenting with rules will not avoid failure, but will drastically reduce the probability of immense failures for an entire region. Thus, a major reason to protect institutional diversity is to reduce the costs of failure when policies are imposed on entire regions without taking into account their diverse

ecological, social, and economic structures. This is particularly important when it comes to the analysis of natural resources – many of which are common-pool resources.

COMMON-POOL RESOURCE PROBLEMS

Many important environmental goods for which uniform solutions are proposed are common-pool resources, which will be the focus for this lecture. *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 (e.g., harvesting a truckload of forest products or withdrawing water from an irrigation system) reduces the resource units that are available to others (V. Ostrom and E. 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 (Myers and Worm, 2003; Mullan *et al*, 2005; FAO, 2005).

In this lecture, 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, or by a self-organised group, as relevant for situations of a particular type. 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 (Crawford and Ostrom, 1995). In light of the rules, and shared norms when relevant, individuals adopt strategies leading to consequences for themselves and

for others. As individuals learn more about the outcome of their own and other's actions within a particular situation, they may change norms and strategies, leading to better or worse outcomes for themselves and the relevant environment.

One of the earliest, most powerful, and long-lasting model of a common-pool resource is the static model of a fishery published by Scott Gordon in 1954. In 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 an equilibrium where individual revenue equalled their cost. Achieving this individually profitable level of harvesting, however, wastes substantial resources and threatens the long-run sustainability of the resource. More and more harvesters want to enter the resource, and eventually they can destroy it.

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: 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 that is needed is for a government to impose rules so that harvesters face different incentives and withdraw at a maximum sustained yield.

RECOMMENDING OPTIMAL INSTITUTIONS

The widespread acceptance of the Gordon model 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 (Terborgh, 1999, 2000; Lovejoy, 2006), 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).

GOVERNMENT PROPERTY

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). More than 100,000 protected areas already exist and include approximately 10 per cent of the forested areas in the world (Barber *et al*, 2004). While considerable enthusiasm exists for creating protected areas, their performance varies 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 (Nepstad *et al*, 2006; Hockings, 2003). 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 more than 200 protected areas in twenty-seven 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). Due 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 of over-time remotely sensed images of deforestation levels in national parks located in the *same* country have found that some are well protected and others were 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.

PRIVATE PROPERTY

Private property is frequently recommended as 'the' way to reduce the tragedy of the commons. And, some private property systems backed up by community and government institutions have worked rather well. In the Los Angeles metropolitan area after World War II, for example, water producers from several groundwater basins used the California courts as an arena in which to determine who had rights to pump how much water per year (see Chapter 4 in Ostrom, 1990). The court established a Watermaster to determine factual information initially needed to determine rights and then to monitor the conformance of water producers to the agreements (Blomquist, 1992). In the groundwater basins that were adjudicated and rights allocated, markets for water rights emerged rapidly. Further, 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 or by ceasing or changing their business) to users who wished to expand their access to local water sources.

After a half-century, 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 (Steed and Blomquist, 2006; Blomquist and Ostrom, 2008). In some years, producers were authorised to take more than their assigned rights so long as they then

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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. Further, 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. They have invested their own tax revenue in developing methods for replenishing the basin with reclaimed sewer water and desalinated ocean water. The water levels in West Basin, which is located adjacent to the Pacific Ocean, have been substantially restored over time and the threat of massive loss due to salt water intrusion has been overcome (Steed, 2010). 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 for creating private property in regard to fisheries (Scott, 1988; Raymond, 2003). 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: 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: 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 program, not of the ITQ system itself, although the latter no doubt implies a degree of acceptance and support of the observer program.)

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.

COMMUNITY PROPERTY

While strong involvement of a community is an important factor in long-run sustainability, community property is also not a panacea (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 (Platteau, 2004; Oyono *et al*, 2005) leading in some cases to the exclusion of the poorest members of a community (Malla, 2000).

Simply turning common-pool resources over to local users by itself is unlikely to avoid overharvesting especially if these same resources were taken away from users decades earlier and are in a degraded condition when returned. Some communities manage their fisheries, water systems, 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.

Over the past decades, colleagues associated with the Workshop in Political Theory and Policy Analysis have conducted studies of water resources and forestry in many countries and have found that when local users do have a significant voice – if not full ownership – over a resource, they can frequently improve outcomes rather significantly. Since many of the successful community-managed resources are the result of the

Table 1: Relationships between governance structure and physical condition of irrigation systems

Physical condition of irrigation systems		Types of governance structure		Chi-square value	Significance
		FMIS (%)	AMIS (%)		
Physical condition	Excellent [37]	18.2	8.4	23.02	0.00
	Moderately good [144]	67.4	45.8		
	Poor [48]	14.4	45.8		
Technical efficiency	Highly efficient [58]	28.9	12.5	27.30	0.00
	Moderately efficient [137]	62.8	50.0		
	Inefficient [33]	8.3	37.5		
Economic efficiency	Highly efficient [66]	33.2	12.5	45.35	0.00
	Moderately efficient [140]	63.5	52.1		
	Inefficient [23]	3.3	35.4		

Note: Number of irrigation systems is in brackets.

Source: Joshi *et al* (2000: 78).

hard work of very poor individuals, these findings are surprising to many in the developed world who think that one always needs extensive financial resources to be successful in managing common-pool resources. An example of successful, but quite diverse forms of community property for sustaining resources is the extensive number of farmer-managed irrigation systems (FMIS) in Nepal.

COMPARING FARMER-MANAGED TO AGENCY-MANAGED IRRIGATION SYSTEMS (AMIS) IN NEPAL

Farmers have survived over the centuries in much of Asia due to their evolved knowledge of how to engineer complex irrigation systems including dams, tunnels, and water diversion structures of varying size and complexity, well matched to local geographic and ecological conditions. None of these systems work well, however, without agreed-upon rules for allocating water as well as allocating responsibilities for providing the needed labour, materials, and money

to build the systems in the first place and maintain them over time. Since Nepal was governed by a collection of princes until 1848, farmers built paddy rice systems through the centuries without a central government that took major responsibility for planning, building, or maintaining these systems. Even when the Rana family consolidated power in the mid-nineteenth century, very little national attention was paid to irrigation until the 1950s. In the mid-1950s, a Department of Irrigation was established and a series of Five Year Plans articulated and developed. Since then, the Asian Development Bank, the World Bank, CARE, the International Labor Organization, and other donors have invested very large sums in designing and constructing large-scale, AMIS in some regions of Nepal.

Thus, farmers in Nepal have long exerted local authority to create their own water associations, construct and maintain their own systems, and monitor and enforce conformance to their rules (see Benjamin *et al*, 1994; Lam *et al*, 1997; Yoder, 1994; Regmi, 2007). The irrigation systems constructed and

maintained by farmers tend to rely on low-tech construction techniques including building non-permanent headworks from mud, trees, and stones. International aid agencies have provided considerable funding to government agencies in an effort to upgrade the engineering standards.

Colleagues associated with the Irrigation Management Systems Study Group at the Institute of Agriculture and Animal Science, Tribhuvan University in Nepal, have been working with colleagues at Indiana University since the early 1990s (Shivakoti *et al*, 1992; Benjamin *et al*, 1994; Lam *et al*, 1994). We have jointly developed the Nepal Irrigation Institutions and Systems (NIIS) database that now has information about 231 irrigation systems located in twenty-nine out of the seventy-five districts in Nepal (Joshi *et al*, 2000; Ostrom *et al*, 2011).²

Our consistent finding, and that of other scholars doing research on irrigation in Nepal (Gautam *et al*, 1992), is that on average, FMIS outperform AMIS on multiple dimensions (Shivakoti and Ostrom, 2002). That farmers have organised an irrigation system is the variable with the largest explanatory power of any that we have identified in the NIIS studies. Let me provide a very brief overview of our findings from this extended research.³

Focusing on three measures of the physical condition of the irrigation system at the time of data collection, as shown in Table 1, a larger proportion of FMIS are able to maintain the overall physical condition of the system in excellent or moderately good condition as contrasted to AMIS, as well as achieving higher technical and economic efficiency (see Lam 1998 for definitions of these concepts). The better physical condition of the canals enables FMIS to achieve increased levels of cropping intensity at both the head and tail end of a canal, as shown in Table 2. Thus, the investment of farmers in keeping their systems in good physical condition pays off in regard to significantly more agricultural productivity.

About two-thirds of both FMIS and AMIS have formal written rules that include provisions for imposing fines on farmers for not contributing resources to operate and manage the systems (Joshi *et al*, 2000: 75). On the other hand, in eight out of ten AMIS, an official guard is hired, while only six out of ten FMIS rely on an official guard (*ibid.*). The presence of an official guard, however, does not translate into an increased likelihood that fines will actually be imposed. On 75 per cent of the FMIS, fines are actually imposed when farmers are observed to break a rule, while fines

Table 2: Relationships between governance structure and cropping intensity of irrigation systems

Cropping intensity		Types of governance structure		Chi-square value	Significance
		FMIS (%)	AMIS (%)		
Intensity at head end	High [142]	70.2	52.2	5.27	0.02
	Low [72]	29.8	47.8		
Intensity at tail end	High [123]	65.1	34.1	13.74	0.00
	Low [87]	34.9	65.9		

Note: Number of irrigation systems is in brackets.

Source: Joshi *et al* (2000: 80).

are actually imposed on only 38 per cent of the AMIS (ibid.: 76). Farmers follow the rules of their system to a greater extent on FMIS than on the AMIS and they also tend to achieve a higher level of mutual trust (ibid.).

The specific rules that the farmers use in governing their systems on a day-to-day basis vary substantially from one system to another. The 'official' guard on many of these systems is one of the farmers themselves who 'rotates' into this position on a regular basis. The rules specifying allocation rules, responsibilities for monitoring, and punishment, however, are not consistent from one system to the next. Thus, the monitoring of water allocation and contributions to maintenance is largely performed by farmers who have participated in the crafting of the specific rules of their own system and have a strong interest in seeing their system perform well and ensure that others on the system are not free-riding or taking more water than their official share. These findings raise policy-relevant questions about the value of centralised and capital-intensive strategies for providing irrigation. They also confirm the importance of two design principles originally identified by Ostrom (1990): proportionality in benefits and costs, and collective-choice arrangements that involve individuals affected by the resource system.

Thus, farmers with long-term property rights, who can communicate, develop their own agreements, establish the positions of monitors, and sanction those who do not conform to their own rules, are likely to grow more rice, distribute water more equitably, and keep their systems in better repair than is the case in government systems. Since many of the government systems rely on high-tech engineering, the capability of farmers to increase agricultural production on their 'primitive systems' while they also provide the labour to maintain

and operate the system is particularly noteworthy.

In a recent book, several colleagues examine the process and impact of an innovative irrigation assistance project that was undertaken in Sindhu Palchok in the mid-1980s under the imaginative leadership of Prachanda Pradhan and Robert Yoder (Ostrom *et al*, 2011). Using Qualitative Comparative Analysis and other statistical and qualitative methods, we found that the initial and later investments in system infrastructure are but one factor that may lead to longer-term success – but not simply that investment in infrastructure by itself, as has been so often recommended in the development literature. We found that unless the farmers organise themselves and create their own rules, and augment their rules through collective action or by imposing fines on those who violate rules, infrastructure investment *alone*, while potentially outcome enhancing in the short run, is not sufficient for achieving sustainable higher performance.

We also learn that there is a difference between how rules are defined and operationalised in diverse settings. In his study of irrigation systems in Nepal, Shukla (2002) found that almost all of the systems he studied had well-demarcated boundaries. A substantial difference existed, however, between FMIS as contrasted to AMIS. On the FMIS, the farmers themselves determine how large the area to be served should be. The farmers who demarcate the boundary will also have to participate in the construction of the system and its maintenance by contributing time, materials, and potentially some funds. Thus, the boundaries of irrigation systems developed by farmers tend to be conservative so that those who make the system work are more assured of getting water.

The boundaries of AMIS, by contrast, are frequently demarcated as part of donor-funded projects. Irrigation engineers are

strongly motivated to show a positive benefit–cost ratio. The more farmers placed within the service boundary of a system, the higher the benefits that can be reported in the plans submitted to donors for funding. Once funding is granted, few efforts are made to check the reliability of earlier estimates. Farmers in the larger service area are promised water, but may not receive a reliable supply. Farmers on these systems are more likely to steal water and less likely to contribute resources to maintenance.

The study of irrigation systems in Nepal is only one of the empirical studies we have undertaken over the past quarter of a century focusing on institutional arrangements and their impact on incentives, behaviour, and outcomes. I will now provide a brief overview of our research related to forest resources and institutions.

STUDYING FORESTS AROUND THE WORLD

A long-term collaborative research network – the International Forestry Resources and Institutions (IFRI) research programme – was established in the early 1990s with centres now located in Bolivia, Colombia, Guatemala, India, Kenya, Mexico, Nepal, Tanzania, Thailand, Uganda, and the United States, with a new centre being established in Ethiopia (see Gibson *et al*, 2000; Poteete and Ostrom, 2004; Wollenberg *et al*, 2007; <http://www.sitemaker.umich.edu/ifri/home>). IFRI is unique among efforts to study forests, as it is the only interdisciplinary long-term research programme studying forests owned by governments, by private organisations, and by communities in multiple countries.

In an effort to examine whether government ownership of protected areas is a necessary condition for improving forest

density, Hayes (2006) used IFRI data to compare the rating of forest density (on a five-point scale) assigned to a forest by the forester or ecologist who had supervised the forest mensuration of trees, shrubs, and groundcover in a random sample of forest plots.⁴ Of the 163 forests included in the analysis, seventy-six were government-owned forests *legally designated* as *protected forests* and eighty-seven were public, private, or communally owned forested lands used for a diversity of purposes. No statistical difference existed between the forest density in officially designated protected areas versus other forested areas. Our early studies focused on outcomes achieved by differently organised forests at one time period (see Agrawal, 2001; Agrawal and Ostrom, 2001; Gibson *et al*, 2005).

We have now been able to return to some of our forest sites for a second or third visit (see Gautam *et al*, 2004; Nagendra *et al*, 2005; Nagendra, 2007). Chhatre and Agrawal (2008) have examined the changes in the condition of 152 forests under diverse governance arrangements as affected by the size of the forest, collective action around forests related to improvement activities, size of the user group, and the dependence of local users on a forest. They found that ‘forests with a higher probability of regeneration are likely to be small to medium in size with low levels of subsistence dependence, low commercial value, high levels of local enforcement, and strong collective action for improving the quality of the forest’ (ibid.: 1327). In a second major analysis, Chhatre and Agrawal (2009) focus on factors that affect tradeoffs and synergies between the level of carbon storage in forests and their contributions to livelihoods. They find that larger forests are more effective in enhancing carbon and livelihoods outcomes, particularly when local communities also have high levels of

rule-making autonomy. A very recent study by Persha *et al* (2011) that examines the potential for social-ecological synergy again finds that local participation in forest governance (in diverse formal governance systems) is strongly associated with positive outcomes for the users and for the ecology. Recent studies by Coleman (2009) and Coleman and Steed (2009) also find that a major variable affecting forest conditions is the investment by local users in monitoring. Further, when local users are given harvesting rights, they are more likely to monitor illegal uses themselves. Many other focused studies also stress the relationship between local monitoring and better forest conditions (Ghate and Nagendra, 2005; Nagendra, 2007, 2008).

The legal designation of a forest as a protected area is *not* by itself related to forest density. But detailed field studies of monitoring and enforcement, as they are conducted on the ground, illustrate the challenge of achieving high levels of forest regrowth without active involvement of local users (see Batistella *et al*, 2003; Agrawal, 2005; Andersson *et al*, 2006; Tucker, 2008). Our research shows that forests under different property regimes – government, private, communal – sometimes meet enhanced social goals such as biodiversity protection, carbon storage, or improved livelihoods. But at other times, these property regimes fail to provide such goals (Dietz *et al*, 2003). Indeed, when governments adopt top-down decentralisation policies that leave local officials and users in the dark, stable forests may become subject to deforestation (Banana and Gombya-Ssembajjwe, 2000; Banana *et al*, 2007). Thus, it is not the general type of forest governance that is crucial in explaining forest conditions; rather, it is how a particular governance arrangement fits the local ecology, how specific rules are developed and adapted over time, and whether users consider the system to be

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legitimate and equitable (for a more detailed overview of the IFRI research programme, see Chapter 5 in Poteete *et al*, 2010).

To conclude our brief overview of research related to community involvement in the governance of forests (including direct community ownership, government concessions, or other long-term co-management arrangements), we have shown communities to be as effective or, under certain conditions, more effective than government ownership (Bray *et al*, 2005). The debate over the effectiveness of 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.

Some public policies have misunderstood the difference between self-organised systems and centralised government policies to 'decentralise' the governance of a resource. We find a variety of outcomes when forest resources have been 'decentralised' in a centralised manner (Agrawal and Gupta, 2005; Agrawal and Ostrom, 2008; Webb and Shivakoti, 2008). As discussed above, we find that when forest users have a voice in the

design of the rules they will be using related to forest and other resources, they can frequently devise rules well matched to the complexity of the ecological system involved (Gautam, 2007). Simple solutions do not exist, however, for managing complex ecologies (Campbell *et al*, 2006; McPeak *et al*, 2006). Thus, our research illustrates that enabling resource users to have a significant voice in the governance of natural resources can lead to sustainable outcomes, but we must be careful not to presume that there is a simple way to 'decentralise' the governance of resources using a single formula for an entire region or nation.

FROM OPTIMAL SOLUTIONS TO ADAPTIVE MULTILEVEL 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.

Institutional theorists need to recognise what ecologists recognised long ago: the complexity of what we study and the necessity of recognising the nonlinear, 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: 2) has summarised:

That is, ecosystems are complex, adaptive systems and hence, are characterized by historical dependency, complex

'... the same rules that work well in one setting are part of failed systems elsewhere!'

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 theorists need to recognise that deriving a simply 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 multilevel (Wilson, 2006; Wilson *et al*, 2007). We also need to draw on research using multiple methods to examine relationships among social and ecological structures and outcomes (Poteete *et al*, 2010). 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.

We need to think about complex adaptive systems as being composed of a large number of active elements whose rich patterns of interactions produce emergent properties that are not easy to predict by analysing the separate parts

of a system. Holland (1995: 10) views complex adaptive systems as 'systems composed of interacting agents described in terms of rules. These agents adapt by changing their rules as experience accumulates'. Complex adaptive systems 'exhibit coherence under change, via conditional action and anticipation, and they do so without central direction' (ibid.: 38–39). Systems that do not have central direction can take on many forms. One form that is frequently proposed is a completely decentralised layer of governments that are composed entirely of self-organised, local bodies each of which govern a particular smaller-scale resource system. Our own research supports more complex, adaptive designs that do enable the users to have a substantial voice in the design and monitoring of the rules in use but also involve larger units in a polycentric system (V. Ostrom, 1999).

THINKING ABOUT POLICY RECOMMENDATIONS

In earlier efforts to analyse which rules worked best related to fisheries, irrigation systems, and forests, we have found a simply gigantic number of individual rules that were used in the field (Tang, 1994; Schlager, 1994; Ostrom, 2005). The search for rules to improve the efficiency, sustainability, and equity of the outcomes obtained in common-pool resources is an incredibly complex task involving the consideration of a potentially infinite combination of specific rules that could be adopted. A key problem of all common-pool problems, for example, is that of excluding free riders. Thus, an essential set of rules defines who is eligible to use a common-pool resource.

Empirical studies of boundary rules used in field settings have identified twenty-seven different types of boundary rules used alone or in combination

'Our own research supports more complex, adaptive designs that do enable the users to have a substantial voice ...'.

(Ostrom, 2005; Ostrom *et al*, 1994). A second essential set of rules relates to the particular appropriate and other activities that individuals may take related to a particular commons. Again, empirical studies have identified more than 100 different types of rules that specify when, where, how, and how much of the products of a commons may be appropriated by someone who is authorised to do so. When you consider rules related to incentives and sanctions, to information conditions, and to procedural requirements, the number of rules that could be used to regulate activities related to any particular common-pool resource is very large. Since rules are used in combination with one another, the potential configuration of rules that could be used to improve the efficiency, equity, and sustainability of common-pool resources approaches infinity.

Consequently, instead of assuming that the choice of institutional rules to improve the performance of human systems that utilise common-pool resources – or any other complex task for that matter – is a process of designing optimal rules, we need to understand the policy design process as involving an effort to tinker with a large number of component parts (see Jacob, 1977). Those who tinker with any tools – including rules – try to find combinations that work together more effectively than other combinations. 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. Further, 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.

In all of our studies, we have not yet found *specific* rules that have a statistically positive relationship to performance in a large number of common-pool resources (Gibson *et al*, 2000; NRC, 2002; Dietz *et al*, 2003). 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 for 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 (Weinstein, 2000; Trawick, 2001; Marshall, 2005; Dayton-Johnson, 2000; Sarker and Itoh, 2001; Cox *et al*, 2010).

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 should be encouraged to address rather than as a blueprint they should apply everywhere. 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:

- How are we going to define the physical boundaries of this resource over time?
- Who will be allowed to harvest which kinds of resource units?
- What will be the timing, quantity, location, and technology used for harvesting?
- Who will be 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 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 multilevel, experimental approach rather than a top-down approach to the design of effective institutions.

EXPERIMENTING WITH RULE CHANGES

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 countries with diverse ecologies, 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 Brock and Carpenter, 2007; Berkes, 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 Landau, 1969, 1973; Bendor, 1985; 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).

Notes

1 I deeply appreciate being asked to give the Stein Rokkan Lecture this spring. In this lecture, I have drawn on research on institutions and natural resources conducted over the last five decades. In addition to new work, I have drawn on several earlier papers and I appreciate the support for the research on which this lecture is based that has been provided by the National Science Foundation, the Ford Foundation, and the MacArthur Foundation. Many colleagues at the Workshop in Political Theory and Policy Analysis at Indiana University have contributed to this research through the years as well as colleagues in Africa, Asia, and Latin America with whom I have had the privilege to work. I deeply appreciate Patty Lezotte's creative efforts to help me with the editing of this article.

2 The findings from Nepal discussed in this article are based on data, most of which was collected in earlier, more peaceful times.

3 Readers who wish to dig deeper are encouraged to read Lam (1998), Joshi *et al* (2000), Shivakoti and Ostrom (2002), and Ostrom *et al* (2011, and the extensive references cited therein).

4 Extensive forest mensuration is conducted at every IFRI site at the same time that information is obtained about forest users, their activities and organisation, and about governance arrangements. Comparing forest measures across ecological zones is misleading since the average diameter at breast height in a forest is strongly affected by precipitation, soils, elevation, and other factors that vary dramatically across ecological zones. Thus, we ask the forester or ecologist who has just supervised the collection of forest data to rate the forest on a five-point scale from very sparse to very abundant.

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